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Habitat rehabilitation for inland fisheries

Global review of effectiveness and guidance for rehabilitation of freshwater ecosystems

Cover photos:
Left: excavation of new off-channel habitat (River Drau, Austria)
Top right: lake habitat enhancement in North America
Bottom right: planting of riparian trees (River Drau, Austria)
(Courtesy of Tennessee Wildlife Resources Agency)

FAO Fisheries Technical Paper No. 484
Habitat rehabilitation for inland fisheries.
Global review of effectiveness and guidance for rehabilitation of freshwater ecosystems.

CORRIGENDA

The attached pages should replace pages 11, 43, 44, 68 and 75 that contain printing errors.

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FIGURE 2a

Total number of references from different regions located and used in this review

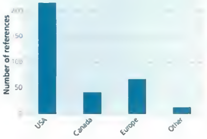
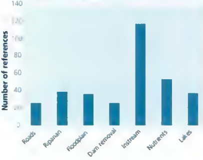


FIGURE 2b

Total number of those references in each category that evaluated project effectiveness



Much of the research on the impact of roads and techniques for rehabilitation has focused on unpaved or gravel roads in forested, mountainous, or rural areas. However, roads and other impervious surfaces have a dramatic effect on watershed processes in urban areas as well. For example, in urban areas roads and other impervious surface areas lead to increases in frequency and magnitude of peak flows, channel incision, and simplification of instream habitat (Scott *et al.*, 1986; Booth, 1990; Wong and Chen, 1993; Moscrip and Montgomery, 1997). When roads and impervious surfaces exceed 10 percent of the total watershed area, changes in hydrology and negative impacts on aquatic habitat and biota will occur (Klein, 1979; Booth and Jackson, 1997).

Road crossings such as culverts can also prevent or inhibit the upstream or downstream migration of many fishes and other aquatic organisms (Larinier, 2002c; Roni *et al.*, 2002). The amount of stream habitat made inaccessible because of human infrastructure is daunting. There are an estimated 1.4 million stream-road crossings in the United States and an estimated 2.5 million artificial barriers prevent fish passage in the United States (US Fish and Wildlife, National Fish Passage Programme, Arlington, Virginia, unpublished data). While massive efforts have focused on replacing impassible culverts in streams in the United States and Canada, fish passage through culverts and other road crossings is a problem throughout North America (e.g. USGAO, 2001; Langill and Zamora, 2002; Warren and Pardew, 1998), Europe (e.g. Yanes *et al.*, 1995; Glen, 2002; Larinier, 2002c), and elsewhere in the world.

2.1.1 Techniques for reducing road impacts

A number of rehabilitation actions have been attempted to mitigate for the negative impacts that roads have on aquatic ecosystems. These can be divided into three categories: sediment reduction, restoration of hydrology, and connectivity. Connectivity is extensively discussed in the floodplain rehabilitation section, but here we will discuss it in terms of road crossings of small streams.

A variety of methods are used to reduce sediment delivery and landslides induced by roads or road construction. These include resurfacing roads, reducing traffic, increasing the number of stream crossings, stabilizing cut and fill slopes, and replacing stream crossings to improve the natural transport of sediment and biota (Table 4; Figure 3). In addition, complete road removal or abandonment of roads, which may use many of the previously mentioned techniques, is also a common method to reduce road impacts.

Actions to reduce hydrologic effects of roads in rural or forested areas include some of the same activities as sediment reduction as well as increasing the number of cross drain structures, water bars to distribute water onto the forest floor or into

Table 10

Summary of studies reviewed that examined adult response to instream habitat improvement.

Study	Techniques	Observations
Crispin *et al.*,1993	Various wood and boulder weirs and structures throughout entire anadromous reach	Coho spawner abundance and redds throughout stream increased four fold in years following treatment compared pretreatment (11 years of monitoring)
House, 1984	Gabions and rock weirs	Increased steelhead and coho spawner abundance, and percentage of entire spawner population using treated stream reaches (3 years of monitoring)
House and Boehne, 1985; House, 1996	Gabion weir placement	2.5 fold increase in coho spawner abundance (12 years of monitoring)
Gortz, 1998	Gravel, boulders, and stream deflectors	Five fold in restored versus control reaches (1 year of monitoring)
Moreau, 1984	Boulder weirs, deflectors, and clusters	Documented steelhead and Chinook use of gravels trapped by placed structures. (No preproject data, 1 year of monitoring)
West, 1984	Gravel cleaning, boulder placement	Chinook salmon spawner use increased up to three fold in some areas (two years of monitoring)
Clarke and Scruton, 2002	Gravel additions	Initial results showed spawner use of gravels but monitoring was discontinued before conclusive results were obtained (2 years of monitoring)
Avery, 1996	Gravel riffles and sediment traps	No change in age 0 brook or brown trout (sediment traps rapidly filled with sand, 7 years of monitoring)
Klassen and Northcote, 1988	Gabions weirs	No difference in pink salmon (*Oncorhynchus gorbuscha*) egg to fry survival between gabions and natural sites (1 year of data)

times as many brown trout redds in restore reaches of the River Esrom, Denmark, but found low numbers of fry, suggesting the spawning was not successful. Vehanen *et al.* (2003) found that adult grayling preferred a section of a Finish stream enhanced with boulder and cobble structures (reefs and islands). Iversen *et al.* (1993) reported increased fry and trout numbers in rivers Gudenå and River Hjortvad in Denmark, though they suggested projects on other Danish streams were not as successful because gravel was placed in slow water areas or because of siltation or poor water quality. In some cases the creation of spawning riffles is coupled with sediment traps (e.g. Avery, 1996), which has shown some level of success, but it is unclear how long the sediment traps will function before needing to be reexcavated.

This lack of rigorous evaluation of adult salmon and trout response to instream enhancement stems in part from the multiple generations and thus long time frame (>10 years) needed to detect an adult response to habitat alterations (Bisson *et al.*, 1997; Korman and Higgins 1997). Nonetheless, evaluating adult response is critical for projects focusing on enhancement of spawning habitat and review of the limited published studies suggests increases in adult use of treated areas and increased fry (Table 10).

Nonsalmonid fish response—Less information exists on the response of nonsalmonid fishes to instream habitat enhancement (Bilby *et al.*, 1996; Lonzarich and Quinn, 1995; Roni 2003) (Table 11). A review of habitat enhancement projects in warm-water streams throughout the Midwest found that a wide variety of techniques were successful in altering stream morphology and increasing stream cover, although rigorous physical and biological evaluation of individual projects was generally lacking (Lyons and Courtney, 1990). Lonzarich and Quinn (1995) found no effect of woody debris cover and depth on threespine stickleback (*Gasterosteus aculeatus*) or coast range sculpin (*C. aleuticus*) habitat use, growth, or survival in an artificial stream channel. Roni (2003) examined response of reticulate sculpin (*Cottus perplexus*), torrent sculpin (*C. rhotheus*), larval lamprey (*Lampetra* spp.), and Pacific giant salamanders (*Dicamptodon ensatus*) to LWD placement in the coastal Pacific Northwest of the US and found higher densities of lamprey in treated stream reaches, but little difference for other nonsalmonid species.

Other studies on nonsalmonid fishes have examined changes in species diversity. Shields *et al.* (1993, 1995b) found large increases in fish species diversity following boulder weir placement in two streams in Mississippi (USA). Angermeier and Karr (1984) examined responses of 10 warm-water fishes to wood removal and placement in a small stream in Illinois (USA) and found more and larger fish in stream sections with woody debris. Jungwirth *et al.* (1995) examined instream structures on fishes in seven Austrian rivers and found increases in both fish biomass and species diversity when treated sections were compared with natural sections. Pretty *et al.* (2003) found no improvement in fish species abundance or diversity in 13 English streams treated with flow deflectors or artificial riffles. However, bullhead (*Cottus gobio*) and stone loach (*Barbatula barbatula*) tended to be more abundant in rehabilitated riffles than control areas.

Monitoring the response of the entire fish community is important in determining both project effectiveness and whether instream techniques are restoring fish

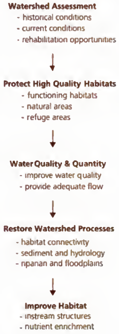
TABLE 11

Studies examining response of non-salmonids or entire fish community to placement of instream structure.

Study	Techniques Examined	Diversity/Richness	Other
Lyons and Courtney, 1990 (reviewed 20 studies)	Various boulder, and bank rotection structures; channel meandering	Three studies reported increase in fish	Nine studies reported increase in abundance (17 studies examined fish)
Roni, 2003	Various wood placement techniques	No difference	Increase in juvenile lamprey, no response for sculpin or salamanders
Shields *et al.*, 1993	Boulder weir placement to aggrade incised channel	Doubled	Approximate tenfold increase in fish biomass after treatment
Shields *et al.*, 1995a,b	Boulder weir placement to aggrade incised channel	Increased	Threefold increase in abundance and median fish size increased
Pretty *et al.*, 2003	Deflectors and stone riffles	No difference between rehabilitated and control	No difference in total abundance though bullhead and stone loach increased slightly
Angermeier and Kerr, 1984	Placement of woody debris	No difference	Increased abundance, and size
Lonzarich and Quinn, 1995	Cover placement in artificial channel	Higher in pools with structure	Higher survival of water column species in pools with cover, but no difference in growth among species
Jungwirth *et al.*,1995	Removal of bank protection and installation of groins and bedfalls	Increased	Increased abundance and biomass
Linlokken, 1997	Rock weirs	No change	Percentage of minnows and brown trout increased, bullheads decreased

FIGURE 13

Strategy for prioritizing watershed rehabilitation actions. First step (box) is to conduct assessments necessary to determine potential opportunities.



4.2 PRIORITIZING ACTIONS

Once the previously described assessments have been completed and a list of potential actions has been established there are several approaches for prioritizing rehabilitation actions depending on the level of information available. Based on our understanding of watershed processes, basic needs of aquatic biota, and our review of effectives of techniques, we recommend an interim approach for setting rehabilitation priorities. This begins first with protecting high quality habitats and providing adequate water quality and quantity and is then followed either sequentially or concurrently by restoring connectivity of habitats, restoring processes that create and maintain habitat, and finally, if necessary, instream habitat improvements (Figure 13). We begin this section by discussing this interim approach and then outlining other approaches for prioritizing rehabilitation.

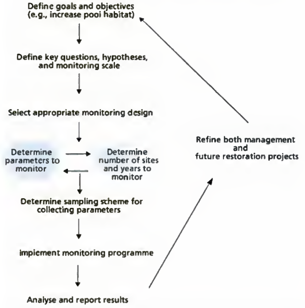
Habitat protection is typically not seen as rehabilitation. However, given the loss of habitat and the continued degradation from human activities, protecting high quality functioning habitats should be part of the rehabilitation planning process (NRC, 1992; Roni *et al.*, 2002; Lucchetti *et al.*, 2005). As discussed in previous sections, it is also considered more cost-effective and reliable than trying to rehabilitate habitat once it has been degraded.

Adequate water quality and quantity are fundamental to the success of any rehabilitation action or suite of actions. Our review of published data and anecdotal evidence suggests that inadequate water quality and quantity have prevented many expensive rehabilitation projects from achieving successful increases in fish production. In some cases adequate water quality can be achieved through riparian protection or replanting, such as in the case of agricultural runoff (Sovell *et al.*, 2000; Meals and Hopkins, 2002; Parkyn *et al.*, 2003). However, pollutants often originate from both point and nonpoint sources including runoff from roads, treated or untreated sewage effluent, or runoff from agricultural lands and will require much more complicated measures to improve water quality. Water quantity is typically addressed by limiting the timing and volume of water withdrawals or water use. Lengthy political, legal, or legislative acts may be needed to restore water quantity and quality. While both water quality and quantity can be difficult to address, they are critical to the success of other rehabilitation actions.

Assuming problems with water quality and quantity have been or are being addressed; we suggest that restoring connectivity of habitats is the next factor to consider. Reconnection of isolated habitats relies on existing habitat and often produces both rapid and long-term results. This includes both restoring longitudinal connectivity such as removal or providing passage at manmade barriers as well lateral connectivity such as reconnecting isolated floodplain habitats and floodplains. The techniques discussed in the section on floodplain connectivity and the culvert/fish

FIGURE 16

Key steps for developing a monitoring and evaluation programme for rehabilitation actions. Modified from Roni (2005).


Define goals and objectives
(e.g., increase pool habitat)
Define key questions, hypotheses,
and monitoring scale
Select appropriate monitoring design
Determine parameters to monitor
Determine number of sites and years to monitor
Determine sampling scheme for collecting parameters
Implement monitoring programme
Analyse and report results
Refine both management and future restoration projects

determining the appropriate scale requires differentiating between changes in local abundance and changes in population parameters at a watershed or larger scale. Most research on habitat and biota, both for rehabilitation and other ecological studies, has focused on reach scale or individual habitat units. This information is important, but uncertainty about movement, survival, and population dynamics of biota prevent these reach-scale studies from addressing watershed or population-level questions. Studies designed to assess watershed or population-level effects can provide valuable information but also face multiple challenges (e.g. upstream-downstream trends, sampling logistics; Conquest 2000; Downes *et al.*, 2002).

From the key questions and specific hypotheses will flow the other important decisions including appropriate monitoring design, duration and scale of monitoring, sampling protocols, etc. The most difficult part and the biggest shortcoming of many rehabilitation evaluation programmes is the study design. As noted in our review of riparian rehabilitation, lack of preproject data, adequate treatments and controls, reference sites, and various management factors have limited the ability of many studies to determine the effects of rehabilitation actions. There are many potential study designs for monitoring single or multiple rehabilitation actions. None is ideal for all situations and each has its own strengths and weaknesses. Hicks *et al.* (1991) distilled these possibilities down to a handful of experimental designs based on whether data are collected before and after treatment (before-after, or post-treatment designs) and whether they are spatially replicated or involved single or multiple sites (intensive or

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Preparation of this document

This technical paper is part of the publication programme of the Inland Water Resources and Aquaculture Service (FIRI) of the Fisheries Department of the Food and Agriculture Organization of the United Nations. It provides a synthesis of information on the effectiveness of habitat rehabilitation for inland fisheries as well as recommendations for implementing, monitoring and evaluating habitat rehabilitation activities. The document is intended to assist managers, practitioners and scientists involved in the restoration of aquatic ecosystems. It draws on the more than 50 years of practical experience of the authors in the field of fisheries and riparian ecology, aquaculture and habitat restoration.

Production of this document was coordinated by Phil Roni under the supervision of Devin M. Bartley (FIRI) through the FAO Visiting Experts Partnership Programme which provided partial support for Dr Roni while working on the project. The United States National Oceanic and Atmospheric Administration (NOAA) Northwest Fisheries Science Center also donated much of Dr Roni's time for the project. Layout and design were by Ms Sylviane Borghesi (FIRI).

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Abstract

The degradation of inland aquatic habitats through decades of human activities has lead to massive efforts to rehabilitate freshwater habitats for fisheries and aquatic resources in watersheds throughout the world. Many texts have been written on techniques for rehabilitation though no comprehensive worldwide review of the effectiveness of techniques has been undertaken. This paper reviews published evaluations of freshwater habitat rehabilitation projects, including studies on roads improvements and sediment reduction, riparian and floodplain rehabilitation, placement of habitat structures in lakes and streams, addition of nutrients to increase aquatic production and other less common techniques. In particular, the authors summarize what is known about the effects of various techniques for restoring natural processes, improving habitat, and increasing fish and biotic production. Recommendations on limitations of techniques, which techniques are effective, as well as information on planning, prioritizing and monitoring rehabilitation projects are also provided.

Despite locating more than 330 studies on effectiveness, as well as hundreds of other papers on rehabilitation, it was difficult to draw firm conclusions about many specific techniques because of the limited information provided on physical habitat, biota and costs, as well as the short duration and scope of most published evaluations. However, techniques such as reconnection of isolated habitats, rehabilitation of floodplains and placement of instream structures have proven effective for improving habitat and increasing local fish abundance under many circumstances. Techniques that restore processes, such as riparian rehabilitation, sediment reduction methods (road improvements), dam removal and restoration of floods, also show promise but may take years or decades before a change in fish or other biota is evident. Other techniques such as bank protection, beaver removal and bank debrushing can produce positive effects for some species but more often produce negative impacts on biota or disrupt natural processes.

Comparing the cost-effectiveness of different types of rehabilitation techniques was not possible because few evaluations reported various costs or economic benefits; however, estimates of average costs for various techniques are provided. Monitoring and evaluations clearly need to be designed as part of the rehabilitation action. The authors discuss the key steps to consider when designing monitoring and evaluation of rehabilitation actions at various scales.

Similar to less-comprehensive reviews of rehabilitation, this review demonstrates three key areas lacking in most rehabilitation projects: 1) adequate assessment of historic conditions, impaired ecosystem processes and factors limiting biotic production; 2) understanding upstream or watershed-scale factors that may influence effectiveness of reach or localized rehabilitation; and 3) well-designed and -funded monitoring and evaluation. These are the same factors that consistently limit the ability of published studies to determine the success of a given technique at improving habitat conditions or fisheries resources. Finally, this review suggests that many habitat rehabilitation techniques show promise, but most have not received adequate planning, monitoring or cost-benefit analysis.

Key words: habitat rehabilitation, restoration, fisheries, riparian, floodplain, monitoring and evaluation

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Acronyms and abbreviations

BACI	Before-after control-impact
CICFRI	Central Indian Capture Fisheries Research Institute
DVWK	Deutscher Verband für Wasserwirtschaft und Kulturbau
FISRWG	Federal Interagency Stream Restoration Working Group (US)
LIFE	Financial Instrument for the Environment (European Union)
LWD	Large woody debris
NOAA	National Oceanic and Atmospheric Administration
NRC	National Research Council (US)
NRRS	National River Restoration Synthesis Project
RSPB	Royal Society for Protection of Birds (United Kingdom)
USA	United States of America
USGAO	United Stated General Accounting Office

1. Introduction

The degradation of aquatic habitats from human activities has been occurring for centuries, but became particularly pronounced in the late nineteenth and twentieth century in Europe, the United States of America, Australia, and other developed countries (Brookes, 1992; Welcomme, 1994; Arthington and Pusey, 2003). In most developing countries modification, degradation, and pollution of aquatic habitats began in the late twentieth century and continues today (Parish, 2004; Welcomme and Petr, 2004a). Estimates from throughout the globe suggest that 75 to 95 percent of riverine habitats are degraded (Benke, 1990; Dynesius and Nilsson, 1994; Muhar *et al.*, 2000). Dynesius and Nilsson (1994) estimated that 77 percent of the 139 largest rivers (discharge > $350m^3/s$) in the Northern Hemisphere have modified hydrologic regimes. Inventories for specific countries are even higher. For example, 80 percent of large rivers in Austria are moderately to heavily impacted by human development (Muhar *et al.*, 2000) and it is estimated that only 2 percent of the river kilometres in the United States of America are pristine (Benke, 1990).

Human activities such as forestry, agriculture, channelization, power production, industrialization, water use and many others can have a variety of negative consequences for fisheries and aquatic resources. Forest management practices have negatively impacted many streams by increasing fine and coarse sediment, altering stream hydrology, disrupting delivery of woody and organic debris, and simplifying habitat (Meehan, 1991; Murphy, 1995; Erskine and Webb, 2003). Improving the navigation of rivers and estuaries through dredging and snagging (removal of wood) has greatly simplified many rivers and is still widely practiced today (Sedell and Froggatt, 1984; Buijse *et al.*, 2002; Collins *et al.*, 2003). Agricultural activities have had detrimental effects on estuaries, floodplains, wetlands, and low-gradient tributaries through dredging, draining, filling, pollution, channelization of waterways, and diversion of water for irrigation (NRC, 1992; Cowx and Welcomme, 1998; Welcomme and Petr, 2004a). Irrigation and the over appropriation of stream flows particularly in arid regions has led to low stream flows, higher water temperatures, reduced total wetted habitat, reduced ability of the stream to transport sediment, and other deleterious effects (Orth, 1987; Hill *et al.*, 1991; World Commission on Dams, 2000; Parish, 2004). Mining and other extraction industries have had many negative effects on streams from direct alteration and removal of substrates to pollution and release of toxic substances (Nelson *et al.*, 1991). Residential development, industrialization, and urbanization have lead to a suite of problems for aquatic habitats including filling and channelization, changes in hydrology from increased impervious surface area, pollutants from point and nonpoint sources, elimination of riparian zones, and simplification of habitat (Booth, 1990; Booth *et al.*, 2002; Riley, 1998; Konrad, 2003). All these factors have contributed to the degradation and simplification of inland aquatic habitats across entire ecosystems as well as the loss of biodiversity and are the basis for the development of numerous rehabilitation techniques.

Sustainable inland fisheries depend on well functioning ecosystems and adequate habitat. The importance of fishery habitat is acknowledged in the FAO Code of Conduct for Responsible Fisheries which was adopted by the FAO Conference "to ensure sustainable exploitation of aquatic living resources in harmony with the environment" (FAO 1995). Articles 2g) and 6.1of the Code call on States to, "promote protection of living aquatic resources and their environments..." and "... conserve aquatic ecosystems", respectively. Article 6.8 of the Code specifically addresses habitat rehabilitation: "All critical fisheries habitats ... should be protected and rehabilitated as far as possible and where necessary". A review of the effectiveness of habitat rehabilitation efforts is in Article 7.6.7, "In the evaluation of alternative[1] conservation and management measures, their cost-effectiveness ... should be considered", and Article 7.6.8, "The efficacy of conservation and management measures should be kept under continuous review." FAO recognizes that conservation and management strategies must be viewed realistically taking into account numerous alterations and perturbations to inland water ecosystems and recommended that strategies should avoid "vain attempts to restore a substantially altered ecological balance" (FAO, 1997).

In response to the degradation of aquatic habitats from a variety of human activities, rehabilitation of aquatic habitats has become commonplace throughout developed countries and is increasing in developing countries (NRC, 1992; Cowx and Welcomme, 1998). These efforts are often undertaken to restore or improve natural resources that are of economic, cultural, or spiritual importance. Rehabilitation efforts typically occur throughout a watershed including both riparian and upland activities as well as activities in the lowlands such as reconnection of floodplains and addition of habitat structures (e.g. logs, boulders, weirs) in streams. The vast majority of these efforts have been undertaken to restore fisheries resources and in some cases large sums of money are spent on a single species or group of species. For example, hundreds of millions of dollars are spent annually in western North America in an effort to increase Pacific salmon (*Oncorhynchus* spp.) runs that once sustained large commercial and sport fisheries but are now threatened with extinction. Other ecosystem restoration programmes have been initiated in the Florida Everglades, the Missouri, Mississippi, and Sacramento rivers, the Louisiana Delta, Chesapeake Bay, the Great Lakes and other major basins (Northeast Midwest Institute, unpublished data; http://www.nemw.org/restoration_summary.htm). Similar efforts are underway in Europe to rehabilitate and reconnect habitats throughout large river basins such as the Rhine and Danube as well as many other rivers and water bodies (Buijse *et al.*, 2002). Many are funded through the European Union's environmental programs (i.e. LIFE or The Financial Instrument for the Environment). Interest in inland fisheries rehabilitation is increasing in many developing countries not only a result of declines in fisheries resources, but also because of desertification in arid regions owing to over appropriation of stream flows or increased flooding in temperate and tropical areas due to poor land use practices (Parish *et al.*, 2004). For example, large efforts are underway to reforest areas and restore floodplain wetlands in the Yangtze River basin in China to reduce flooding, while efforts are underway in the arid Timar River basin to restore steamflows and wetlands and halt desertification and loss of biodiversity (Parish, 2004). Similarly, reflooding large portions of the Mesopotamian Marshes in Iraq, which almost disappeared following extensive draining in the 1990s, is currently underway (Richardson *et al.*, 2005).

Restoration ecology is a relatively young, interdisciplinary field and the literature on aquatic rehabilitation is extensive yet somewhat fragmented (Buijse *et al.*, 2002). Several existing publications that discuss different techniques for rehabilitation in North America (Hunter, 1991; NRC, 1992; Hunt, 1993; Slaney and Zoldakas, 1997; FISRWG, 1998), Europe (Brookes and Shields, 1996; Petts and Calow, 1996; RSPB *et al.*, 1994; Cowx and Welcomme, 1998; Vivash, 1999), Australia (Rutherfurd *et al.*, 2000), and east Asia (Parish *et al.*, 2004) are applicable to many regions throughout the world. Other texts discuss the ecological basis for restoration (e.g. Naiman and Bilby, 1996; Calow and Petts, 1994; Perrow and Davy, 2002) and some regional papers or grey literature discuss effectiveness of different techniques (e.g. Binns, 1999; Roni *et al.*, 2002; Avery, 2004). Unfortunately, no comprehensive review of the effectiveness of various rehabilitation techniques has been completed. In particular, the effectiveness of these efforts at restoring natural watershed processes, improving fish habitat, and increasing fish and biota abundance, and the cost and benefits of various techniques needs to be examined (Roni *et al.*, 2002).

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[1] Habitat rehabilitation is a form of fishery management and conservation. It is unclear what the Code means by "alternative" measures.

The goals of this document are to synthesize the available information on the effectiveness of various habitat rehabilitation techniques and provide guidance for restoration of aquatic ecosystems. By outlining the shortcomings of techniques as well as our understanding of aquatic systems, we strive to improve our understanding of how to plan and prioritize projects as well as to monitor and evaluate their success. To provide some common background, we begin with a general discussion of rehabilitation and restoration terminology and provide an overview of ecological processes that create aquatic habitats and their importance in planning rehabilitation projects. Next we synthesize what is known about the effectiveness of different categories of techniques at restoring natural processes, improving physical habitat, and increasing fish and biotic production. We then discuss the costs and cost-effectiveness of various activities, and provide recommendations for prioritizing restoration as well as monitoring and evaluation. We focus on freshwater habitats and habitat modifications and while other factors are important for successful watershed rehabilitation, such as remediation of pollutants, toxicology, water quality and quantity, we do not discuss them here. Rehabilitation of estuarine and marine habitats, an equally large field, will be covered in another volume.

1.1 HABITAT RESTORATION VS. REHABILITATION

There are many different definitions of restoration and rehabilitation and practitioners and researchers are in disagreement as to what constitutes restoration (Gore, 1985; Cairns, 1988; NRC, 1992; Kauffman *et al.*, 1997). The term restoration, which in the most formal sense is returning an ecosystem to its original predisturbance state, has commonly been used to refer to all types of habitat manipulations including enhancement, improvement, mitigation, habitat creation, and other situations (Table 1). These activities are more accurately termed rehabilitation, as most do not truly restore a system and in many areas were the land use is predominantly agricultural, residential, urban or industrial, true restoration is not feasible in the foreseeable future (Stanford *et*

TABLE 1
Commonly used terminology and general definitions*

Term	Definition
Restoration	To return an aquatic system or habitat to its original, undisturbed state
Rehabilitation	To restore or improve some aspects or an ecosystem but not necessarily to fully restore all components
Habitat Enhancement or Improvement	To improve the quality of a habitat through direct manipulation (e.g. placement of instream structures, addition of nutrients)
Reclamation	To return an area to its previous habitat type but not necessarily to restore fully all functions (e.g. removal of fill to expose historic estuary, removal of a levee to allow river to periodically inundate a historic wetland)
Mitigation	Actions taken other than habitat rehabilitation to alleviate or compensate for potentially adverse effects on aquatic habitat that have been modified or lost through human activity (e.g. creation of new wetlands to replace those lost by a land development)

* The authors use the term rehabilitation because it includes both full restoration (which is difficult to achieve in many populated areas) as well as the other activities described below. Modified from Roni (2005).

al., 1996). Therefore, we use the term habitat rehabilitation throughout this document to refer to the various activities and where appropriate use more specific terminology (see Table 1).

1.2 ECOLOGICAL BASIS FOR REHABILITATION

River rehabilitation has often been pursued with little knowledge of the natural structure and function of rivers, and projects have commonly attempted to create habitats considered suitable for a single species of interest (Frissell and Nawa, 1992; Muhar *et al.*, 1995; Ward *et al.*, 2001; Roni *et al.*, 2002; Ormerrod, 2004). However, many scientists have pointed out that worldwide declines of fishes and other aquatic species in fresh waters are partly a result of trying to manage individual species and habitat characteristics rather than managing whole ecosystems (e.g. Doppelt *et al.*, 1993; Muhar *et al.*, 1995; Muhar, 1996; Frissell *et al.*, 1997). Scientists and managers alike have also recognized that rehabilitation actions are more likely to be successful at restoring individual or multiple species and preventing the demise of others if they are considered in the context of the surrounding watershed or ecosystem (Doppelt *et al.*, 1993, Muhar *et al.*, 1995, Reeves *et al.*, 1995, Beechie and Bolton, 1999; Habersack, 2000). The watershed and ecosystem contexts are also critical to understanding the effectiveness of various practices, as well as why they succeed or fail (Harper *et al.*, 1998, Jungwirth *et al.*, 2002). International organizations have also adopted an ecosystem approach to fisheries development and conservation. The Convention on Biological Diversity (CBD) (UNEP, 1998) and FAO (FAO, 1995; Sinclair and Valdimarsson, 2003) have recognized the necessity of, *inter alia*, understanding ecological processes, the bio-physical-chemical qualities of aquatic habitat, nutrient cycling, the importance of non-target species, predator-prey relationships and even the role of humans in ecosystems in order to ensure long-term survival of aquatic species. Therefore, before discussing the effectiveness of habitat rehabilitation techniques, some background on watershed processes is necessary to assist in understanding the variability in effectiveness among techniques as well as among studies on a given rehabilitation method.

Effective planning, implementation, and evaluation of rehabilitation actions requires assessment of disrupted ecosystem functions that reduce the productivity of river systems and are responsible for declines in aquatic ecosystem integrity (Beechie and Bolton, 1999; Buijse *et al.*, 2002). The goal of such assessments is to identify alterations of key processes that affect stream habitats and specify management actions required to restore or rehabilitate those processes that sustain aquatic habitats and support biological integrity (e.g. Muhar *et al.*, 1995; Tockner *et al.*, 1999; Buijse *et al.*, 2002; Beechie *et al.*, 2003a). In this approach, restoring specific fish populations (or populations of any other single organism) is subordinate to the goal of restoring the ecosystem that supports multiple species. As long as all rehabilitation actions are consistent with the overriding goal of restoring ecosystem processes and functions, habitats will be restored for multiple species.

The scientific basis for this approach can be summarized in two important characteristics of biota and their habitats (adapted from Beechie and Bolton, 1999):

1. Biota are adapted to local environmental conditions.
2. Spatial and temporal variations in landscape processes create a dynamic mosaic of habitat conditions in a river network.

These statements imply that biota are adapted to spatially and temporally variable habitats, and suggest that such environmental variability is important to the long-term survival of fish stocks or races. Thus it does not make sense to manage for the same conditions in all locations, or to expect conditions to remain constant in any single location. This has been recognized in scientific critiques of many management issues in the past decade, including "one-size-fits-all" habitat standards (Muhar *et al.*, 1995; Bisson *et al.*, 1997), not managing for spatial or temporal variation in habitats

(Reeves *et al.*, 1995; Bisson *et al.*, 1997; Habersack, 2000), and addressing symptoms of a disrupted ecosystem rather than the causes (Frissell and Nawa, 1992; Muhar *et al.*, 1995; Spence *et al.*, 1996; Ormerrod 2004). Such approaches generally do not consider that local populations are adapted to the natural potential habitat conditions within their range, and that those conditions vary in space and time. By contrast, identifying the root causes of degradation (i.e. impaired ecosystem processes and functions) focuses rehabilitation actions on those processes that form and sustain habitats. This allows each part of the river network to express its natural potential habitat, and helps conserve and restore the natural spatial and temporal variation of habitats to which aquatic communities are adapted (Muhar, 1996; Beechie and Bolton, 1999).

We stress that identifying the root causes of ecosystem degradation is important for two main reasons. First, our understanding of most of the linkages between landscapes, habitat, and aquatic biota is fraught with uncertainty, and we cannot predict exactly how land uses alter aquatic habitat conditions or how those habitat changes alter biota. In fact, it can be argued that we are not yet even aware of all the aspects of aquatic ecosystems that significantly affect fish populations. This lack of knowledge has in the past led to significant habitat degradation. For example, the role of wood debris in habitat formation was poorly understood until the 1970s, and removal of wood and channelization of rivers over the past 200 years resulted in dramatic alteration of river habitats (Sedell and Luchessa, 1982; Muhar *et al.*, 1995; Collins and Montgomery, 2002). As recently as the 1980s biologists recommended widespread wood removal for habitat improvement, not recognizing its importance to the structure and function of aquatic ecosystems. Today wood removal is far less common (but still occurs), however, the example serves to illustrate that we could have avoided significant habitat loss by choosing management actions that preserved riparian forest processes and natural wood functions in channels, even without understanding the value of wood in aquatic ecosystems.

Second, traditional rehabilitation actions such as boulder and log structures or spawning gravel placement attempt to build habitats that do not move in space or time, whereas natural habitats are typically created by movement of river channels, wood debris, and sediment. Therefore, many rehabilitation actions fail to restore habitats because they do not recognize the integrated nature of physical and ecological processes in watersheds (Frissell and Nawa, 1992; Ward *et al.*, 2001; Ormerrod, 2004). As we shall see in subsequent sections of this document, this relative lack of knowledge leads to two main types of failure: 1) site-prescribed engineering solutions can be overwhelmed by altered watershed processes that are far removed from degraded habitats (e.g. increased sediment supply from upslope sources can bury instream structures and pools), or 2) such measures can prevent habitat formation that would otherwise naturally occur (e.g. bank protection prevents formation of new floodplain habitats). Avoiding these types of project failure requires that we focus on restoring ecosystem processes and functions that form and sustain aquatic habitats, rather than on the habitats themselves. Understanding these factors will also help the reader to understand project successes and failures discussed in subsequent sections of this report.

Beechie *et al.* (2003b) proposed a simple conceptual framework detailing relationships among ecosystem processes, rehabilitation actions, habitat conditions, and biota (Figure 1). Using this framework we discuss actions that restore either ecosystem processes (e.g. sediment reduction actions, riparian planting, floodplain rehabilitation) or connectivity of habitats (e.g. restoring fish passage) and contrast those with rehabilitation actions that directly manipulate or modify stream habitats (e.g. instream structures, bank hardening, construction of floodplain ponds). We recognize that most evaluations of habitat rehabilitation have focused on instream habitats, with a lesser degree of emphasis on connectivity. There have been relatively few evaluations of the effectiveness of actions to restore sediment supply or hydrologic regime, in part because these processes are naturally highly variable and relatively small changes are hard to detect at the watershed scale. Moreover, such processes often require decades for results to manifest in stream channels, making linkage of such actions to changes in habitat characteristics and biota even more difficult (Beechie *et al.*, 2005).

River ecosystems throughout the world are driven by the same fundamental sets of processes, such as supply of water, sediment, and organic matter (Leopold *et al.*, 1964; Welcomme, 1985; Knighton, 1998). However, the specific mechanisms driving each of the inputs vary depending on geologic, topographic, and climatic setting (Table 2). Therefore, restoration or rehabilitation actions will address the same sets of processes, but techniques will vary depending on the specific mechanisms that must be addressed. For example, sediment supply to the stream network should be considered in any watershed, but certain processes of sediment supply may be emphasized depending on location. Sediment supply is dominated by landslides in humid mountainous regions (Sidle *et al.*, 1985), but is more commonly a function of surface erosion and gullying in gentler terrain or semiarid regions (Dunne and Leopold, 1978; Darby and Simon, 1999). Land uses that alter these erosion processes also vary by region, and techniques needed to address changes in sediment supply must be adapted to specific erosion processes and land use impacts.

Conducting watershed assessments and understanding important processes driving habitat development and degradation are critical elements in planning rehabilitation actions. Identifying these processes can help determine the success of the various stream and lake rehabilitation techniques we will discuss.

FIGURE 1

Diagram of linkages between landscape controls, habitat-forming processes, habitat conditions and biological responses. (Source: Beechie *et al.*, 2003b)

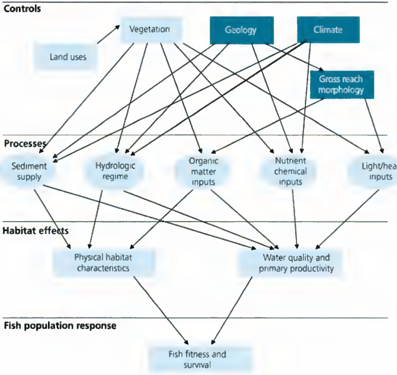


TABLE 2

Examples of variation in dominant ecosystem processes and human impacts across ecological settings. Assessments should target those processes that are locally important within each region or watershed.

Watershed process or function	Ecological setting	
	Semiarid, gentle terrain	**Humid mountains**
Sediment	Gullying and surface erosion, agriculture impacts	Mass wasting and gullying, forestry impacts
Hydrology	Flashy flow regime, diversions and dams	Rainfall to snowmelt floodplain regimes, dams
Riparian functions	Grasses and shrubs retain sediment, grazing and farming impacts	Forest shades streams and supplies organic matter, logging impacts
Habitat connectivity	Culverts, dams, and dikes common	Culverts, dams, and dikes common

2. Effectiveness of common techniques

Dozens of methods and techniques have been developed to rehabilitate freshwater habitats, ranging from methods that try to restore natural processes (e.g. riparian replanting, sediment reduction) to those that seek to create immediate changes in physical habitat (e.g. placement of instream habitat structures) in hopes of creating rapid increases in target species. Typically many types of rehabilitation are undertaken at the same time at a given site or within a catchment. We categorize these rehabilitation activities based on the general area where they occur or the processes they seek to restore including: 1) roads and uplands, 2) riparian, 3) floodplain, 4) dam removal, 5) instream structures, 6) structures in lakes and ponds, 7) nutrient enrichment, and 8) miscellaneous activities (e.g., habitat protection, bank protection, beaver reintroduction or removal) (Table 3). We briefly describe each type of rehabilitation and then describe what is known about its effectiveness at improving natural processes (delivery of wood, water, sediment, nutrients, etc.), physical habitat, and biota. For biotic responses we focus primarily on fishes and to a lesser extent on macroinvertebrates, but report on plants and other biota when available and appropriate. Detailed information on specific techniques and their designs can be found in RSBP *et al.* (1994) Cowx and Welcomme (1998), Slaney and Zoldakas (1997), FISRWG (1998), and Vivash (1999).

To assess the effectiveness of rehabilitation we conducted an extensive review of the existing literature focusing primarily on peer reviewed literature, supplementing that with information from grey literature when available. We utilized databases such as Aquatic Fisheries and Science Abstracts and Web of Science to locate relevant papers, books, and technical reports on restoration theory (general background information) and, more importantly, evaluations of project effectiveness. We also searched library catalogues of the United Nations Food and Agriculture Organization, the US National Oceanic and Atmospheric Administration's Northwest Fisheries Science Center, and the University of Washington, as well as conducted Internet searches with relevant keywords (e.g. aquatic restoration, rehabilitation, habitat improvement). We obtained more than 700 papers on freshwater habitat rehabilitation and restoration including 334 that reported results of scientific evaluations of the effectiveness of one or more habitat rehabilitation techniques (Figure 2a). There were also many other books and reference material examined. The vast majority of the studies on both rehabilitation and effectiveness were from the United States, Canada and Western Europe, with a relatively small number from other countries (Figure 2b). This likely reflects the size of the economy, funding for research *programmes*, and money spent on habitat rehabilitation in these developed countries versus elsewhere in the world. Most of the published literature on effectiveness focuses on instream rehabilitation or nutrient enrichment, while most of the general or theoretical literature focuses on floodplain restoration or more general references on various types of restoration. Thus the reader needs to be conscious that the following discussion on the effectiveness of various techniques is biased as it is limited by the amount of published information available on effectiveness for a given techniques.

TABLE 3

Common categories of habitat rehabilitation discussed in this review and examples of common actions. More detailed descriptions of techniques are provided in the section on effectiveness.

Category	Examples of common techniques	Typical Goals
Road rehabilitation	• Removal or abandonment • Resurfacing • Stabilization • Addition or removal of culverts	• Reduce sediment supply • Restore hydrology • Improve water quality
Riparian rehabilitation	• Fencing to exclude livestock • Removal of grazing • Planting of trees and vegetation • Thinning or removal of understory	• Restore riparian vegetation and processes • Improve bank stability & instream conditions
Floodplain connectivity	• Levee removal • Reconnection of sloughs, lakes • Excavation of new floodplain habitats • Remeandering a straightened stream	• Reconnect lateral habitats • Allow natural migration of channel
Dam removal and flood flows	• Removal or breaching of dam • Increase instream flows • Restoration of natural flood regime	• Reconnect migration corridors • Allow natural transport of sediment and nutrients
Instream habitat structures	• Placement of log or boulder structure • Engineered log jams • Placement of spawning gravel • Placement of brush or other cover	• Improve instream habitat conditions for fish
Lakes habitat enhancement	• Placement of logs and brush • Artificial reefs • Addition of spawning gravels	• Provide cover, rearing, and spawning habitat
Nutrient enrichment	• Addition of organic and inorganic nutrients	• Boost productivity of system to improve biotic production • Compensate for reduced nutrient levels from lack of anadromous fishes
Miscellaneous techniques	• Reintroduce or remove beaver • Brush removal • Bank protection	• Reduce or increase habitat complexity • Prevent erosion or channel migration
	• Habitat protection through land acquisition, conservation, easements, or legal protection (laws) • Instream flows	• Protect habitat from further degradation • Provide adequate flows for aquatic biota and habitat

2.1 ROAD REHABILITATION

The construction of both paved and unpaved roads can have a number of negative impacts on aquatic ecosystems (Trombulak and Frissell, 2000; Gucinski *et al.*, 2001). Roads alter hydrologic regimes (Harr *et al.*, 1975; King and Tennyson, 1984; LaMarche and Lettenmaier, 2001) and sediment supply to streams (e.g. Sidle *et al.*, 1985), which influence channel and habitat characteristics (e.g. Cederholm *et al.*, 1982; Hicks *et al.*, 1991; Nyssen *et al.*, 2002) and ultimately impact aquatic biota (Waters, 1995; Gucinski *et al.*, 2001; Gibson *et al.*, 2005). Forest road drainage connections to stream channels (e.g. ditches draining to streams) can alter the amount and timing of water delivery to streams, as well as the delivery of sediment eroded from hill slopes or road surfaces (Croke and Mockler, 2001; Madej, 2001). Roads also alter sediment supply through increased frequency of landslides (Dyrness, 1967; Megahan and Kidd, 1972; Sidle *et al.*, 1985; Best *et al.*, 1995) and increased surface erosion (Reid and Dunne, 1984; Bilby *et al.*, 1989). Ultimately, the changes in sediment and hydrology have a negative impact on many fishes and other aquatic biota.

FIGURE 2a
Total number of references from different regions located and used in this review

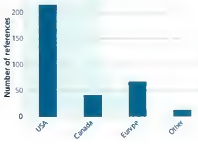
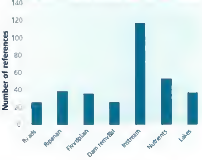


FIGURE 2b
Total number of those references in each category that evaluated project effectiveness



Much of the research on the impact of roads and techniques for rehabilitation has focused on unpaved or gravel roads in forested, mountainous, or rural areas. However, roads and other impervious surfaces have a dramatic effect on watershed processes in urban areas as well. For example, in urban areas roads and other impervious surface areas lead to increases in frequency and magnitude of peak flows, channel incision, and simplification of instream habitat (Scott *et al.*, 1986; Booth, 1990; Wong and Chen, 1993; Moscrip and Montgomery, 1997). When roads and impervious surfaces exceed 10 percent of the total watershed area, changes in hydrology and negative impacts on aquatic habitat and biota will occur (Klein, 1979; Booth and Jackson, 1997).

Road crossings such as culverts can also prevent or inhibit the upstream or downstream migration of many fishes and other aquatic organisms (Larinier, 2002c; Roni *et al.*, 2002). The amount of stream habitat made inaccessible because of human infrastructure is daunting. There are an estimated 1.4 million stream-road crossings in the United States and an estimated 2.5 million artificial barriers prevent fish passage in the United States (US Fish and Wildlife, National Fish Passage Programme, Arlington, Virginia, unpublished data). While massive efforts have focused on replacing impassible culverts in streams in the United States and Canada, fish passage through culverts and other road crossings is a problem throughout North America (e.g. USGAO, 2001; Langill and Zamora, 2002; Warren and Pardew, 1998), Europe (e.g. Yanes *et al.*, 1995; Glen, 2002; Larinier, 2002c), and elsewhere in the world.

2.1.1 Techniques for reducing road impacts

A number of rehabilitation actions have been attempted to mitigate for the negative impacts that roads have on aquatic ecosystems. These can be divided into three categories: sediment reduction, restoration of hydrology, and connectivity. Connectivity is extensively discussed in the floodplain rehabilitation section, but here we will discuss it in terms of road crossings of small streams.

A variety of methods are used to reduce sediment delivery and landslides induced by roads or road construction. These include resurfacing roads, reducing traffic, increasing the number of stream crossings, stabilizing cut and fill slopes, and replacing stream crossings to improve the natural transport of sediment and biota (Table 4; Figure 3). In addition, complete road removal or abandonment of roads, which may use many of the previously mentioned techniques, is also a common method to reduce road impacts.

Actions to reduce hydrologic effects of roads in rural or forested areas include some of the same activities as sediment reduction as well as increasing the number of cross drain structures, water bars to distribute water onto the forest floor or into existing stream channels, and other techniques to prevent the road or road ditches as serving as channel networks. Reducing the amount of road surface draining directly to streams can also reduce fine sediment delivery.

In urban areas several new techniques have been applied in recent years to control increased runoff and increase stormwater infiltration. Those include but are not limited to porous paving, removing hard surfaces where possible and planting with vegetation, a variety of detention, retention, or seepage basins or ponds, overflow wetlands, addition of rainwater cisterns, high-flow bypass channels, alternative drainage systems, and low-impact development for new construction (Sieker and Klein, 1998; Riley, 1998) (Figure 4). Much of the work on urban systems focuses on protecting waterways in suburban or rapidly urbanizing areas by limiting impervious surface areas and protecting riparian zones and uplands. In contrast, work in existing urban centres has focused on preventing further degradation and structural improvements to channels and riparian areas when possible (Sailer, 1994; Horner and May, 1999).

Culverts, bridges, and fish migration ladders are used to restore connectivity among stream reaches and channels at road crossings. Most are designed to provide adequate adult fish passage at road crossings, but not all provide passage for juvenile fishes or maintain natural processes (e.g. sediment and wood transport) and many affect channel morphology (Table 5). Bridges may be costly, but do not constrain the channel as much as culverts and allow the passage of other materials and formation of a natural stream channel. Open bottom culverts or embedded (e.g. countersunk) pipe-arch culverts allow a natural substrate to form within the channel and are effective at passing both juvenile and adult salmonids (Furniss *et al.*, 1991; Clay, 1995; Larinier,

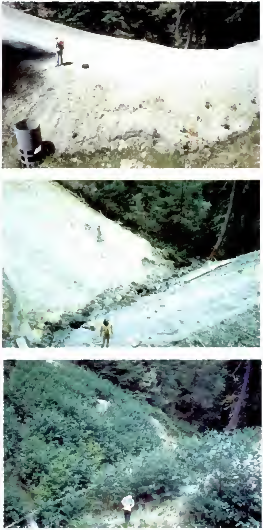

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FIGURE 3. Examples of forest road removal before, immediately after, and a few years after removal of a stream crossing and road fill in Redwood National Park, California.

TABLE 4

Overview of rehabilitation techniques commonly used to reduce sediment inputs and hydrologic effects from roads (Modified from Beechie et ai., 2005). D = road decommissioning, I = road improvement, B = commonly used in both decommissioning and improvement projects. Techniques are described in detail in Reid and Dunne (1984) Bilby *et al.* (1989), Burroughs and King (1989), Chatwin *et al.* (1994), Furniss *et al.* (2000), and Madej (2001).

General objective (process addressed	Site-scale objective	Techniques	Primarily utilized during
Stabilize road (reduce mass wasting)	Remove unstable road material	• Sidecast removal	B
	Reinforce unstable material	• Buttress toe slope	I
		• Retaining walls or other geotechnical approaches	I
	Route water away from unstable material	• Enhance road drainage control	B
		• Subsurface drain pipes or other drainage modifications	I
Protect exposed soil (reduce surface erosion)	Reduce traffic effects	• Block vehicle entry with gate	I
		• Block vehicle entry with barrier (boulders/tank traps)	D
	Armor running surface	• Add rock surfacing to tread	I
		• Pave tread	I
	Vegetate exposed soil surfaces	• Seeding and planting	B
		• "Rip" (i.e. decompact) tread to improve growth of vegetation	D
	Armor exposed soil surfaces	• Cover with rock or other resistant material	B
		• Cover with matting	I
Disconnect road drainage from streams (reduce surface erosion and hydrologic change)	Disconnect road runoff from stream	• Add more cross-drains (e.g. culverts, water bars) between streams	B
		• Outslope tread	B
	Filter sediment from road runoff prior to stream entry	• Install settling ponds	I
		• Install slash filter windrows	B
Reduce drainage diversion (reduce surface erosion and mass wasting)	Reduce fill erosion at stream crossings	• Remove fill at crossing	D
		• Replace soil fill with rock or concrete	I
		• Armor fill surface with rip-rap	I
		• Plant woody species on fill	B
	Improve stream crossings to minimize potential for plugging or diversion	• Replace undersized culverts with adequate structure	I
		• Remove culvert or crossing structure	D
		• Construct drainage dip or hump over structure	I
	Improve road drainage system between stream crossings	• Reshape tread (e.g. inslope, crown)	B
		• Repair or upsize cross drains	I
		• Clear or enlarge ditches	B
	Improve cross-drainage to minimize diversion	• Replace culverts with water bars or dips	D
		• Construct dips or backup water bars over culverts	I

2002c). However, such culverts may constrain the stream channel if the culvert size does not account for large flow events or the volume of sediment and wood transported by the stream (Robison et al, 1999). Other design options include backwatering culverts at the outlet or inlet and placing baffles within the culvert to reduce flow velocity. Clay (1995) and Larinier (2002c) provide concise reviews of culvert designs and methods for retrofitting impassible culverts. While most culverts are designed to pass adult fish, additional research is needed to confirm which types effectively pass juvenile fish at a variety of flows. An intensive review of fish passage structures and their effectiveness for different species is beyond the scope of this document, but can be found in Clay (1995), DVWK (2002) and Larinier (2002a,b).

2.1.2 Effectiveness of techniques for reducing road impacts

With the exception of stream crossings such as bridges and culverts, which can inhibit the migration of fishes and other organisms, all of the research on the effectiveness of road rehabilitation efforts has focused on their physical effects on the stream channel and occasionally on processes such as erosion rates and landslides in mountainous regions. While many of the techniques discussed above are widely utilized in mountainous areas, few published evaluations of their effectiveness exist (Switalski *et al.*, 2004). Evaluations of road-surface erosion reduction techniques have generally been limited to comparisons of fine sediment concentrations in road runoff at different traffic levels and with different surfacing materials. For example, Bilby *et al.* (1989) found a positive relationship between traffic levels and fine sediment delivery to stream channels. Reducing traffic levels in the Clearwater River watershed in Pacific Northwest US reduced surface erosion by a factor of 10 (Reid and Dunne, 1984).

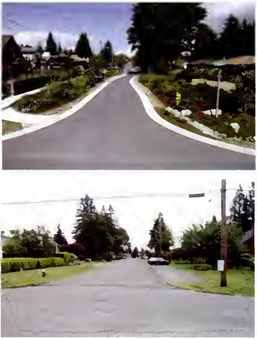
CITY OF SEATTLE PUBLIC UTILITIES

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FIGURE 4. Example of recontouring and reduction of impervious surfaces to reduce storm flows in an urban watershed. Reduction of impervious surface, addition of vegetation, and swales to capture rain and storm water. Preliminary results indicate that project reduced storm flow and runoff from street to stream by more than 95 percent.

Other studies have demonstrated that reducing traffic levels and tire pressure of trucks can reduce sediment delivery from forest roads (Foltz, 1998; Foltz and Elliot, 1998). Reid and Dunne (1984) demonstrated that increasing the thickness of surfacing material to 15.2 cm reduces surface erosion by about 80 percent. Similarly, Kochenderfer and Helvey (1987) demonstrated that surfacing an Appalachian Mountain forest road with 7.6 cm (3 inches) of gravel reduced soil loss from 47 to 6 tons per acre. Burroughs and King (1989) reviewed unpublished studies evaluating different methods for reducing sediment erosion from roads and indicated that surfacing the road with 7.5 to 15 cm of crushed rock greatly reduced erosion. Moreover, reducing truck traffic on unpaved

TABLE 5

Summary of various stream crossing structures and whether they allow for fish passage (juvenile and adult salmonids) and the transport of sediment and large woody debris (LWD) or impact stream morphology by constraining the channel. Modified from Roni *et al.* (2002).

Stream crossing type	Provides fish passage		Transports		Constrains channel[b]
	Adult	Juvenile	Sediment	LWD	
Bridge	Yes	Yes	Yes	Yes	No
Culvert					
Bottomless pipe arch	Yes	Yes	Yes	No	Yes
Squash pipe or countersunk	Yes	Yes	Yes	No	Yes
Round corrugated, baffled	Yes	Yes	No	No	Yes
Round corrugated, no baffles	Yes or no[a]	Yes or no[a]	No	No	Yes
Smooth (round or box)	No[a]	No[a]	No	No	Yes

[a] Fish passage depends on culvert slope and length

[b] Depends on size of culvert or bridge relative to channel and floodplain width

roads or paving of the road surface greatly reduced erosion (Burroughs and King, 1989). These studies, while limited to the western United States, suggest that these methods when implemented properly can reduce surface erosion.

Road removal or abandonment has also been demonstrated to greatly reduce sediment delivery (Madej, 2001; Hickenbottom, 2000; Switalski *et al.*, 2004). For example, Hickenbottom (2000) demonstrated in a Montana watershed that 12 months after road removal and recontouring runoff and erosion approached those of natural slopes. The type of treatment following road removal or closure (e.g. recontouring of slope, ripping of road surface, removal of stream crossing, placement of mulch, seeding, planting) can influence the sediment production and infiltration capacity of the former road bed (Cotts *et al.*, 1991; Maynard and Hill, 1992; McNabb, 1994; Luce, 1997; Elseroad *et al.*, 2003). In general, these studies have found that recontouring slope and site preparation (ripping and mulching of former road bed) are the most effective at inducing plant growth and reducing fine sediment production, though position of the road on the slope and time since treatment also play a role in the effectiveness of these methods.

Very few evaluations of rehabilitation techniques for landslide hazard reduction have been conducted. Harr and Nichols (1993) provided anecdotal evidence that road removal resulted in reduced landslide rates in mountainous areas. Cloyd and Musser (1997) examined a subset of over 1 200 kilometres of treated and untreated forest roads stabilized or obliterated in Oregon and found higher problem rating (erosion and mass failure severity) on untreated roads associated with stream crossings. Changes in turbidity associated with stream crossing and road rehabilitation have occasionally been examined; however, results generally show short-term, construction-related increases in turbidity (Brown, 2002). Finally, it is important to note that the geology plays an important role in the level of natural sediment delivery, the sediment delivery from roads, and the overall success of sediment reduction effort through road resurfacing or complete road removal (Bloom, 1998).

Much of the literature on urban stormwater management and hydrology has focused on modeling of potential changes (e.g. Sieker and Klein, 1998; Johnson and Caldwell 1995) and few published evaluations exist. Booth *et al.* (2002) indicated that stormwater retention ponds have been generally inadequate to alleviate channel erosion or restore hydrology in highly urbanized areas because of the small size of most the ponds. Newer methods for stormwater reduction and reducing hydrologic and water quality impacts through natural drainage systems (e.g. constructed swales, wetlands, vegetated areas) have shown promise, but are in early stages of development and evaluation.

Physical studies on channel response following culvert removal are rarely conducted or published. In a rare study on channel adjustment following culvert removal, Klein (1987) reported channel adjustments following removal of culverts in a California watershed were minimal when large woody debris or other channel roughness elements were present. Most published studies have, however, focused on biotic response or recolonization—usually the major objective of culvert replacement projects. Fish often colonize new habitats relatively quickly and several studies have demonstrated the effectiveness of replacing stream culverts at allowing fish migration (Iversen *et al.*, 1993; Bryant *et al.*, 1999; Glen, 2002; see also section on floodplain rehabilitation). However, if fish numbers are extremely low or if a culvert is only passable at some water levels or seasons, it may take several years for fish to colonize new habitats and monitoring to evaluate success may need to be long term. Studies comparing different types of habitat rehabilitation techniques for salmon have shown removing barriers that block fish migrations leads to some of the largest increases in fish production (Scully *et al.*, 1990).

In addition to benefiting fishes, culvert replacement or other barrier removal projects may benefit invertebrates and wildlife (Yanes *et al.*, 1995; Vaughan, 2002). For example, Yanes *et al.*, (1995) found that culverts both provided and inhibited migration of many mammals and reptiles, and Vaughan (2002) suggested that culverts inhibit upstream movement of many aquatic macroinvertebrates. Finally, the ability of various types of stream crossing to allow for fish and other aquatic organism passage involves a complex relationship between the physical characteristics of the stream crossing (e.g. depth, velocity, roughness) and the behaviour and swimming performance of the fish or other biota (Clay, 1995; Larinier, 2002b,c). While the published literature is limited, it is clear that replacing culverts and other road crossing structures that prevent or inhibit fish migration is a highly successful technique to increase fish habitat and production.

2.1.3 Conclusions – techniques for reducing road impacts

Road removal, abandonment, replacement of impassible culverts and other techniques provide some obvious benefits in terms of sediment reduction, restoring natural hydrologic regimes and habitat connectivity in aquatic systems. There are, however, relatively few published studies on recovery of watershed processes such as hydrology and sediment delivery following road improvements or removal because of the high cost of long term monitoring needed to detect these changes. Little long term monitoring is typically conducted or published on removal of culverts that block fish access because they provide almost immediate and relatively obvious results. Based on our review of the literature on rehabilitation methods to reduce impacts of roads on aquatic systems, we provide the following recommendations with the understanding that new studies are needed to provide more specific recommendations.

- Several techniques (e.g. traffic reduction, resurfacing, removal) have been developed that appear successful at reducing erosion and landslides and improving hydrology associated with roads in forested areas.
- Many techniques exist for reducing impacts of urban stormwater, though their effectiveness is unclear.
- Replacing culverts or other stream crossings that prevent migration of fish is a highly effective technique for restoring connectivity, though stream crossing types vary in their ability to transport sediment and restore other watershed processes.

2.2 RIPARIAN REHABILITATION

Riparian zones are the interface between terrestrial and aquatic environments that provide important chemical, physical, and biologic functions within a watershed, including processing nutrients, delivering woody debris and organic matter to a stream, providing shade, stabilizing soils, regulating microclimate, and many other important functions (Naiman *et al.*, 1993; Pollock *et al.*, 2005). They are also important habitat for both terrestrial and aquatic biota and are zones of particularly high productivity. Unfortunately, because of their proximity to water, transportation and the productivity of the forests and soils, they have been and continue to be highly impacted by anthropogenic activities (Pollock *et al.*, 2005). Common activities in riparian areas include agriculture, grazing, logging, recreation, building of roads and transportation, as well as various residential, urban, and industrial developments (see also section on floodplains). In an effort to improve habitat for fish and wildlife and for a variety of cultural and ecological reasons, large efforts have been initiated to restore riparian areas and improve stream habitat quality.

2.2.1 Techniques for riparian rehabilitation

Several strategies have been developed for either restoring or improving riparian condition typically with the ultimate goal of improving aquatic habitats. These fall into two major categories: 1) silviculture treatments and 2) fencing and grazing reduction. We discuss the two separately as the approaches to rehabilitation of riparian areas are quite different. Silviculture treatments seek to restore riparian areas through replanting of trees and other vegetation, typically with protection from further harvest or vegetation removal. Grazing management or reduction efforts typically seek to remove or limit pressure on riparian areas (grazing) and allow vegetation to recover naturally (sometimes called passive restoration).

2.2.1.1 Riparian silviculture

A variety of silviculture treatments have been used to improve or restore riparian conditions and riparian forests. These include seeding, planting and removal of trees or removal of competing understory vegetation, as well as the removal of a disturbance suppressing vegetation or the alteration of physical conditions (e.g. flood regimes or sediment supply) such that desired vegetation becomes established or undesirable vegetation dies (Smith, 1986; Verry *et al.*, 1999; Stromberg, 2001; Pollock *et al.*, 2005) (Figure 5). Riparian rehabilitation methods can be categorized based on the whether they remove or kill vegetation (thinning/harvest, girdling, competitive release, pruning) or plant vegetation (seeding, planting, coppicing, staking and layering) (Table 6). Most of these techniques were adapted from techniques in upland forests and much of the silvicultural literature focuses on techniques to grow tall, straight, densely packed, commercially important trees, often as monocultures, rather than focusing on creating diverse and functional ecosystems (Smith, 1986; Burkhart *et al.*, 1993; Curtis *et al.*, 1998). There are also a number of nonsilviculture techniques that have been used to improve riparian zones including modifying flow regimes, bank stabilization, and managing invasive exotic plant species.

Because of their natural high rates of disturbance, which create unoccupied patches, riparian areas are relatively susceptible to colonization by exotic species, and in some river systems, the number of exotics is quite high (Nilsson, 1986; deWaal *et al.*, 1995). As such, the problem of controlling exotic species in riparian areas is global in nature (Dudgeon, 1992; Rowlinson *et al.*, 1999; Tickner *et al.*, 2001). However, most of these species do not significantly alter ecosystem functions or out compete native species and thus are not considered invasive or noxious. In contrast, invasive species are aggressive and often form monocultures or become the dominant species if left unchecked. In some instances, riparian areas are dominated by nonnative species and

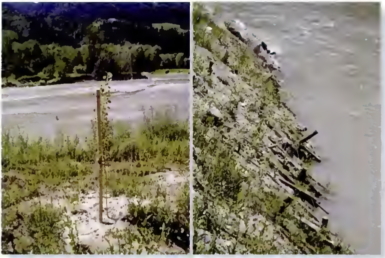

FIGURE 5. Examples of riparian replanting including live trees (left) and planting of willow stakes (right) (River Drau, Austria).

restoration involves removing such species. Tamarisk (*Tamarix ramosissima)* and salt cedar (*T. chinoensis*), invasive species in the American Southwest, for example, have light seeds, enabling them to disperse easily and readily occupy recently disturbed riparian areas, and have caused considerable degradation of riparian habitat throughout the American Southwest (Sudbrock, 1993; Zavaleta, 2000). Other problematic riparian invasive species common in North American and Europe include but are not limited to Japanese knotweed (*Fallopia japonica*), reed canary grass (*Phalaris arundinacea*), giant reed (*Arundo donax*), and leafy spurge (*Euphorbia esula*). There are undoubtedly countless others invasive species in other parts of the world.

Other methods of riparian rehabilitation include more passive efforts such as letting vegetation recover natural or restoration of flood waters. The restoration of flooding or groundwater through dam removal or restoration of normative flows, covered in a later section, is another method for restoring riparian vegetation and river channels

2.2.1.2 Fencing and grazing reduction

Heavy livestock grazing has lead to degradation of riparian and stream habitats throughout the world (e.g. Platts, 1991; NRC, 1992; Robertson and Rowling, 2000; Jansen and Robertson 2001). Riparian areas tend to be more heavily impacted from grazing than upland areas because they are more heavily utilized by livestock and native ungulates owing to their proximity to water and high quality forage (Armour *et al.*, 1991). Livestock grazing can negatively impact riparian and stream environment by changing and reducing vegetation, destabilizing banks, increasing sediment, channel widening, channel aggradation, changing hydrology, and lowering the water table (Armour *et al.*, 1991; Platts, 1991; Belsky *et al.*, 1999). Effects on fish habitat typically include reduction in shade, cover, and terrestrial food supply, increase in water temperature and quality, alteration of stream morphology, and reduction in pools and depth (Armour *et al.*, 1991; Platts, 1991; Belsky *et al.*, 1999).

TABLE 6

Common techniques used in riparian rehabilitation. Pruning and harvest are also occasionally used to improve or restore riparian functions (Modified from Pollock *et al.*, 2005). Dam removal and restoration of floods, which also can improve riparian conditions, are discussed in a separate section.

Technique	Definition	Objectives/comments
Silviculture techniques		
Thinning/ harvest	Cutting or killing of trees in a stand, usually the smaller, less vigorous trees.	Increases the growth rate of remaining trees. Trees can be left on site to provide organic material/LWD to forest floor. Harvest differs from thinning in that larger, commercially valuable trees are taken. Harvest, if done carefully, can minimize damage to riparian functions and in some cases even enhance them.
Girdling	Killing of trees by killing the cambial layer.	Same as thinning, but trees are left standing to create snag habitat and (eventually) LWD.
Competitive release	Killing of vegetation that is competing with desired species (e.g. hardwoods or shrubs competing with conifers).	Increases the growth rate of remaining trees. Can be labour intensive and need to be repeated in riparian areas where growth rates are robust.
Pruning	Removal of limbs from live trees.	Reduces fire hazards or windthrow and allows increased growth of desired species.
Seeding	Planting of seeds of desired species.	Establishes desired vegetation. Often unreliable, depending on species and weather.
Coppicing	Regeneration from vegetative sprouts (stumps, limbs)	Easily establishes vegetation from stumps or limbs (usually certain deciduous trees).
Staking	Vertical insertion of live stems or branches partially into the ground to then take root.	Used to quickly establish trees or bushes. In riparian settings, commonly used to establish willows and cottonwoods.
Layering	Complete or partial horizontal burial of live stems that then take root.	Useful for bank stabilization or other projects where rapid root growth is need. Also used to propagate some conifers that grow slowly.
Planting	Placing live plants of target species into the ground.	Standard technique for establishing plants. Cost of planting largely depends on size of plants.
Site preparation	Alteration of site conditions prior to the application of regeneration techniques.	Done to improve physical conditions so that regeneration, survival rates, and growth rates increase. Burning, dicing, draining, fertilizing, and irrigating are common techniques.
Fencing and grazing reduction		
Removal of grazing	Elimination of livestock from riparian area or entire watershed	To allow natural recovery of riparian area and stream channel, water quality, and biota
Fencing	Placement of fencing perpendicular or adjacent to stream to exclude livestock from part or all of riparian area	To allow natural recovery of riparian area and stream channel, water quality, and biota while allowing for livestock use outside of exclosure
Grazing system	Controlling the number and distribution of livestock, duration and or season of grazing, or control of forage use	Minimize impacts to and allow for recovery of riparian vegetation, stream channel, water quality and biota while still allowing for livestock use

Riparian and stream channel habitats affected by grazing are restored primarily by completely removing grazing, excluding livestock from part of the riparian zone with fences or other structures, or by implementing a grazing reduction or management system that enables riparian vegetation to recover (Elmore, 1992) (Figure 6). Various grazing management systems have been implemented throughout world; almost all involve control of duration of grazing, ungulate numbers, or both. One particularly common one is rest-rotation grazing management, which includes alternating periods of grazing and nongrazing often rotated on multiple pastures. Other techniques to keep livestock out of riparian areas include providing water sources away from the stream (i.e. nose pumps) or providing salt away from the stream (Platts and Nelson, 1985).

2.2.2 Effectiveness of riparian rehabilitation

2.2.2.1 Effectiveness of riparian silviculture

The effectiveness of riparian techniques has been primarily evaluated through short term (<10 years) examination of vegetation survival and growth and few long term evaluations exist (Jorgensen *et al.*, 2000; Pollock *et al.*, 2005). A number of factors effect the growth and survival of riparian plantings including understory or overstory control and grazing by herbivores. Sweeney *et al.* (2002) examined the success of different oak (*Querus* spp.) species over four years under various riparian treatments and silviculture techniques. Protection from herbivores was the most important factor in determining oak seedling survivorship. Emmingham *et al.* (2000) examined over 30 riparian projects

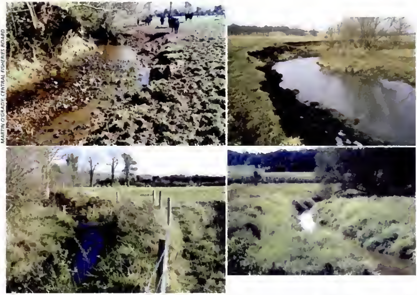
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FIGURE 6. Cartron Stream, County Sligoi, Ireland before (top left) and three years after fencing (bottom left) and Vernon Creek, Wisconsin before (top right) and fifteen years after removal of grazing (bottom right).

designed to convert deciduous riparian areas to coniferous riparian forests in coastal Oregon and suggested that initially riparian silviculture treatments show promise at establishing conifers in hardwood-dominated riparian zones. However, lack of understory and overstory control, protection from browsing by deer (***Odocoileus*** spp.), elk (***Cervus elaphus***), beaver (***Castor canadensis***), and mountain beaver (***Aplodontia rufa***) affected project success. Any rehabilitation activity that involves planting and monitoring vegetation should recognize that herbivores can undermine rehabilitation efforts (Emmingham *et al.*, 2000; Opperman and Merenlender, 2000). Techniques for reducing animal damage to planted vegetation include fencing, plastic mesh cages, netting, and chemical sprays (Black, 1992).

No thorough research on fish response to riparian planting exists and only a few studies have examined other instream biota. Penczak (1995), however, found that the fish species diversity increased in the Warta River, Poland, from 11 to 16 species as riparian vegetation regenerated after removal. A recent examination of riparian fencing and replanting in New Zealand showed that replanted riparian buffers showed improvements in water quality and channel stability, but nutrients and fecal contaminants responses were variable and no improvement in macroinvertebrate communities towards clean water varieties was observed (Parkyn *et al.*, 2003). It suggested that larger or longer buffers were needed to affect changes in water temperature and water quality.

Evaluation efforts for invasive species typically focus on success of removal efforts and success of recolonization by native species (often cottonwoods and willows). Taylor and McDaniel (1998), Roelle and Gladwin (1999), Sprenger *et al.* (2002) and others have monitored and examined the success of various chemical, burning, mechanical, and hydrological treatments in removing tamarisk and the subsequent recolonization of native species. Application of a combination of these techniques in addition to cottonwood and willow plantings and timed irrigations have produced diverse riparian habitat (Taylor and McDaniel, 1998). Roelle and Gladwin (1999) were able to prevent the establishment of tamarisk seedlings at a rehabilitation site by regular, controlled fall flooding. For other invasive species such as Japanese knotweed (Beerling, 1991), which is known to disrupt riparian habitat throughout Europe and North America, there has been little in the way of monitoring to determine at what rate riparian habitat is being lost to the species.

Evidence from the few published studies on passive riparian restoration indicates that natural recovery rates are sometimes sufficient such that hands-on rehabilitation is not necessary and potentially could be counterproductive. Briggs (1996) describes the recovery of Arivaipa Creek, Arizona, after a 500-year flood destroyed most of the riparian vegetation. A massive recovery effort was implemented to accelerate recovery that included planting of thousands of cottonwood stakes. However, just 8 years after the flood, the riparian areas had recovered so robustly with natural vegetation that the artificial plantings could not even be found. Natural recovery of forests is also a common practice to restore seasonally inundated forests that are critical fish spawning and rearing habitat. For example, in Cambodia reforestation is used as an important fisheries rehabilitation technique to create fish spawning, rearing, and feeding areas in seasonally flooded forests adjacent to Tonle Sap (Great Lake) and the Mekong River (Thuok, 1998). Little long term monitoring of natural recovery exists, but it appears to be an effective tool under the right circumstances.

2.2.2.2 Effectiveness of fencing and grazing reduction

The various studies on the effectiveness of grazing reduction methods at restoring riparian areas have examined a broad array of different grazing systems, which makes drawing firm conclusions about the relative effectiveness of a management system other than removal of grazing difficult. Below we summarize what is known about grazing removal or reduction at improving riparian condition, physical habitat, and aquatic biota. This sequence also represents the chronological order in which systems recover following reduction in grazing. When possible we provide examples of effectiveness of different grazing reduction methods.

Riparian conditions—The relatively rapid improvement (5-10 years) of riparian vegetation and riparian functions such as shade, sediment storage, and hydrologic effects (i.e. water storage and aquifer recharge) following livestock exclusion or dramatic reductions in grazing intensity have been documented in several studies (Elmore and Beschta, 1987; Myers and Swanson, 1995; Clary *et al.*, 1996; Kauffman *et al.*, 1997; Clary, 1999; O'Grady *et al.*, 2002). Bank stability, channel geometry, habitat complexity, and other channel characteristics also recover quickly unless the channel is deeply incised (Elmore and Beschta, 1987; Myers and Swanson, 1995).

The most successful strategy for vegetation appears to be complete livestock removal. For example, in a comparison of currently grazed sites and sites where grazing had been removed from 2 to 50 years prior, Robertson and Rowling (2000) found understory vegetation and tree abundance were 1 and 3 orders of magnitude higher, respectively on ungrazed sites. The percentage of bare soil was lower, and the levels of fine coarse and fine particulate organic matter were higher at sites that were still being grazed. Similarly, Platts (1991) reviewed grazing strategies in the US, identified 17 riparian grazing systems, and indicated that light use and complete livestock exclusion provided adequate protection to riparian and fisheries resources. However, the success of rest-rotation grazing systems at allowing vegetation recovery is influenced by many factors including number of days of grazing, season, seasonal livestock dispersal behaviour, and level of compliance (Myers, 1989). Myers (1989) reviewed grazing systems in Montana (USA) and reported that rest-rotation grazing systems that allowed for successful recovery of vegetation averaged 28 days in duration, while unsuccessful averaged 59 days. Platts and Nelson (1985) examined rest-rotation grazing and found compliance during scheduled rest periods was difficult to achieve and cattle grazed less in the riparian areas during the early part of the growing season when high quality upland vegetation was abundant. Historically, grazing systems have not differentiated between riparian and upland range areas (Clary and Webster, 1989), but grazing systems that control the intensity and timing of use of riparian and upland areas are also thought to be beneficial (Elmore, 1992). Spring grazing, for example, can result in equal distribution of stock between riparian and upland areas and maintain herbaceous stubble heights that adequately protect erodable stream banks (Clary and Webster, 1989). Rest-rotation and other seasonal grazing strategies have shown promise at protecting riparian and aquatic habitat when coupled with intensive monitoring (Myers and Swanson, 1995), but these strategies need additional evaluation for effectiveness (Clary and Webster, 1989).

While grazing by native ungulates may have historically degraded some riparian and stream areas, there is evidence that ungulates can help maintain diversity of riparian vegetation (Medina *et al.*, 2005). In a recent study in Scotland, Humphrey and Patterson (2000) found that reintroduction of cattle grazing increased plant community diversity. Conversely, the exclusion of excessive grazing by native herbivores can also assist in the recovery of riparian vegetation. For example, Oppermann and Merenlender (2000) found significantly higher levels of woody plants (*Salix* spp.) along reaches of a California stream where blacktail deer (*Odocoileus hemionus columbianus*) had been excluded with fencing. Moreover, the exclusion of livestock may be followed by an increase in native ungulates, which can also negatively impact efforts for recovery of riparian areas and the stream channel (Medina *et al.*, 2005). These studies suggest that in some case limited grazing by livestock can have some positive effects on riparian vegetation, while grazing of wild ungulates can have some negative consequences.

Physical habitat and channel conditions—Instream habitat conditions recover slower than many riparian conditions (e.g. shade, plant growth); however, several studies in arid environments have reported positive effects of complete grazing removal or fencing on bank stability and in channel features (Platts, 1991; Medina *et al.*, 2005). Myers and Swanson (1995) found that both complete livestock exclusion and rest-rotation grazing increased bank stability, tree cover, and pool habitat, though other management activities (e.g. road crossing, removal of coarse woody debris) negatively impacted instream conditions. Clary *et al.*, (1996) and Clary (1999) reported improved width/depth ratios and substrate embeddedness following complete removal and reduction grazing in studies in both Oregon and Idaho. Connin (1991) reviewed several riparian rehabilitation projects that involved both bank protection and fencing or removing of grazing in the western United States and found in general an increase in bank stability, riparian vegetation growth, and improvements in channel conditions. If livestock are excluded from only a portion of the riparian zone, the width of the exclusion or buffer has been demonstrated to be positively correlated with the level of fine sediment retention (Hook, 2003). These studies demonstrate that grazing systems may also lead to recovery of stream banks, particularly if historic grazing was extremely heavy. Complete grazing removal appears to allow for better recovery of other in-channel factors such as width to depth, channel entrenchment, bank angle, and fine sediment.

Biotic responses—Relatively few published studies exist that examine the effects of grazing reduction on fishes and other aquatic biota (Rinne, 1999). Our understanding of the effects of grazing and grazing reduction on fish populations has been developed primarily through examining the effects of grazing on stream habitat characteristics (Platts, 1991; Clary, 1999). Much of the information is based on studies comparing sites with different levels of grazing and habitat quality and fish numbers, which generally show higher stream temperatures and lower fish numbers in grazed than ungrazed sites or watersheds (e.g. Li *et al.*, 1994; Myers and Swanson, 1991; Platts, 1991; Wu *et al.*, 2000).

Most of the studies on grazing impacts on biota have occurred in the western United States where large tracks of public land are managed for livestock grazing. Platts (1991) and Rinne (1999) conducted thorough reviews on grazing in the western United States and indicated that while some studies demonstrated increases in fish numbers, fish and aquatic biota were rarely the focus of the studies. Rinne (1999) found that most studies were of inadequate experimental design or duration to effectively determine significant changes in fish numbers due to removal or reduction of grazing. In a study in Oregon streams, Kauffman *et al.* (2002) reported improvements in vegetation, stream morphology, and densities of young-of-year rainbow trout (*Oncorhynchus mykiss*) but not adult or juvenile trout following livestock exclusion. They suggested that the lack of response of juvenile and adult fishes was because of the short reaches sampled in their study. Medina *et al.* (2005) reported on three long-term case studies in the American Southwest and found inconclusive results in all three cases due to limitations in study design, species interactions, upstream or watershed-scale effects, limitations of study design, introduction of exotic species, and fisheries management (stocking and changes in fishing regulations).

Few published studies have examined reduction in grazing on fishes or biota in other parts of the world. O'Grady *et al.* (2002), for example, reported an increase in Atlantic salmon (*Salmo salar*), brown trout (*Salmo trutta*), and minnow (*Phobincus phobincus*) numbers following stabilization of banks and fencing in an Irish basin impacted by severe overgrazing. Parkyn *et al.* (2003) found that fencing and planting of riparian buffers produced rapid improvements in water quality and channel stability, but had no detectable effect on macroinvertebrate fauna in New Zealand streams. Similar to North American studies, they indicated that upstream, watershed-scale factors and short buffer reaches likely influenced results for biota. The results of grazing studies from throughout the world emphasize the need for long term, well designed, watershed-scale monitoring that considers many external factors when evaluating grazing reduction.

Other factors–Reduction in grazing can also influence water quality, fish diet, and other biota. In a rare watershed-scale study, Meals and Hopkins (2002) reported that riparian fencing coupled with other bank protection actions reduced P levels up to 20 percent. Similarly, Sovell *et al.* (2000) found lower water quality at continuously versus rest-rotation grazed sites. Removal of grazing can lead to improvements in avifauna most likely through change in riparian community and associated with change in water table (Diaz *et al.*, 1996; Dobkin *et al.*, 1998). Grazing can also affect fish by changing the plant community, which affects fish prey, and in turn fish diet, as demonstrated by Laffaille *et al.* (2000) in a study in a French marsh.

2.2.3 Conclusions – riparian silviculture and fencing and grazing reduction

Different grazing management strategies and silviculture treatments to restore riparian areas and improve fish habitat are common and effective techniques with obvious benefits to riparian vegetation and less obvious benefits to instream factors. Few published evaluations have examined instream factors and many factors can influence the success of different techniques at improving riparian and instream conditions. We provide the following recommendations based primarily on the few published evaluations discussed above and experience.

- The success of planting depends upon a number of factors including site conditions, competition, grazing and browsing, and exotic species.
- Passive restoration of riparian areas may be an effective technique if the disturbance can be removed and invasive species do not compete with native vegetation.
- Complete exclusion and fencing of livestock have shown the most promising results in terms of vegetation, bank, and instream characteristics.
- Rest-rotation or other grazing systems have shown promise under proper management and the right physical, morphological, and climatic conditions.
- Response of stream channel and biota to grazing and riparian silviculture treatments may lag behind vegetation recovery.
- Response of aquatic biota and in-channel conditions, while rarely examined, are influenced by upstream or watershed-scale features and thus may not respond to changes in riparian conditions.
- Several factors may influence success of grazing and riparian rehabilitation techniques including geology, channel type, climate, exotic species, native ungulates, effective control of grazing intensity and duration, size of exclusion or buffer, and upstream processes or impacts.

2.3 FLOODPLAIN REHABILITATION[1]

Floodplain wetlands, lakes, and other aquatic habitat support large fisheries and aquaculture operations in many parts of the world (Welcomme, 1985; Thompson and Hossain, 1998; Hoggarth *et al.*, 1999; Ghosh and Ponniah, 2001; Welcomme and Petr, 2004a,b). The channelization of rivers for industrial, urban, agricultural, and other human uses has lead to a decline in fish production in many floodplain areas (Bayley, 1995; Modde *et al.*, 2001; Pretty *et al.*, 2003; Miranda and Lucas, 2004; Welcomme and Petr, 2004a,b). Habitats such as wetlands, sloughs, and oxbow lakes were once common in large river floodplains, but are now disconnected or rare in many river systems because of channelization and other human activities (Sowl, 1990; Galat *et al.*, 1998).

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[1] Dam removal and restoration of floods can also be considered floodplain rehabilitation; however, we address them in section 2.4 so that we can cover them in detail.

In the last decade, the rehabilitation of large rivers, and reconnection of isolated floodplains and their associated habitats in particular, has become a critical component of river ecosystem rehabilitation in developed countries (Holmes and Nielsen, 1998; Sear *et al.*, 1998; Buijse *et al.*, 2002; Florsheim and Mount, 2002) and more recently in developing countries (Gopal, 2004). Techniques used to reconnect river-floodplain systems and rehabilitate larger streams (> 20m bankfull width) are at the early stage of development (Cowx and Welcomme, 1998), though they have recently been the focus of study on the Danube and Rhine rivers (Buijse *et al.*, 2002) and other rivers in Europe (Iversen *et al.*, 1993; Lusk *et al.*, 2003). Many of the river systems where such rehabilitation efforts have taken or will take place are heavily engineered systems that rely on human intervention to be maintained in a specified state (Bayley *et al.*, 2000; Buijse *et al.*, 2002). Thus many projects and subsequent techniques have been and will be a compromise between allowing natural processes to function and engineering solutions (Cowx and Welcomme, 1998).

2.3.1 Overview of floodplain techniques

The objective of most floodplain or large river rehabilitation efforts is to restore or improve lateral (connection with floodplain), longitudinal (upstream-downstream), and occasionally vertical connections (hyporheic connections). Some of the most common rehabilitation techniques include connection of isolated floodplain habitats (e.g. side channels, ponds, wetlands, excavated ponds), levee removal or setback, and construction of a new channel or realignment or meandering of existing channel or creation of new floodplain ponds (Table 7; Figure 7 and 8). Other techniques include dam removal, restoration of flood flows, and rehabilitation of riparian areas, which are covered in other sections of this document. Floodplain habitats include numerous types of channels, ponds, lakes, and depressions. These areas provide valuable overwintering habitat for fishes that prefer lentic habitats (Peterson and Reid, 1984; Swales and Levings, 1989; Cowx and Welcomme 1998; Buijse *et al.*, 2002). Moreover, in many cases floodplain habitat rehabilitation is recommended over instream habitat improvement in larger channels (>12m bankfull width) and low gradient channels (<2 percent) (Dominquez and Cederholm, 2000; Roni *et al.*, 2002; Pess *et al.*, 2005). In the following subsections we describe various techniques and what is known about their effectiveness. We distinguish between wetland rehabilitation and floodplain rehabilitation by focusing on habitat rehabilitation activities for fish while wetland rehabilitation can include restoration of plants, birds, mammals, and other vertebrates and invertebrates and often focuses on habitat not directly connected with a stream. We view all of these as important goals of habitat rehabilitation, but focus our discussion on fish habitat, rather than purely wetland science (see Kusler and Kentula, 1989, Wheeler *et al.*, 1995, and others for more information on wetlands).

2.3.1.1 Connecting isolated habitats

The level of connectivity of floodplain habitats has a direct effect on fish species utilization and composition (Berrebi-dit-Thomas *et al.*, 2001; Miranda and Lucas, 2004). Reconnecting isolated habitats through weir removal, levee breaching, or replacement of a culvert or stream crossing to reconnect a relic channel or larger floodplain water bodies, such as a pond or lake to the main stem, is one of the more common techniques. Notched dikes, culverts, submersible dikes, and controllable flow gates can be used to reconnect existing relic channels in areas where flow needs to be regulated (Lister and Finnegan, 1997; Cowx and Welcomme, 1998). Two common problems that occur with such projects are: 1) elevation differences between the mainstem and relic channels due to mainstem channel incision from the time of floodplain channel disconnection, and 2) water quality degradation in the floodplain. For example, rehabilitation opportunities identified in the Stillaguamish River in Washington State (USA) include

TABLE 7

Floodplain rehabilitation techniques, modified from Pess *et al.*, (2005). Barrier and dam removal are discussed in sections 2.1 and 2.4, respectively.

Rehabilitation technique	Rehabilitation location	Physical effects	Biological effects	Constraints
Reconnection of floodplain features (e.g. channels, ponds)	Mainstem channels	Flow connection to mainstem	Create rearing environment for fish	Elevation differences, water quality degradation due to land use practices
	Mainstem channels	Increase channel migration; allow for flow, sediment, and nutrient pulses	Increase in habitat types, potential for habitat formation, biological productivity, and diversity	Setback does not allow full floodplain function for entire floodplain
Aggrading mainstem channels (e.g. submersible dams, logjams)	Mainstem channels	Increase channel migration, allow for flow, sediment, and nutrient pulses	Increase in habitat types, potential for habitat formation, biological productivity, and diversity	Works best if floodplain has survived intact
Creation of floodplain habitats	Surface channel	Flow connection to main stem	Create spawning and rearing habitat for fish	Sedimentation problems; does not allow for full floodplain function; In some cases includes bank armoring
	Groundwater channel	Flow connection to main stem	Create spawning and rearing habitat for fish	Does not allow for full floodplain function; in some cases includes bank armoring
	Off-channel ponds from land use (e.g. gravel pits) or alcove pond	Flow connection to mainstem	Creates spawning and rearing habitat for fish	Risk of avulsion, adequate prey resources, cover, and water quality
Barrier or culvert replacement/removal	All channel types	Increase downstream flux of sediment, wood, water, and energy	Increase biotic exchange in up stream and downstream direction	Infrastructure within the mainstem and floodplain may not allow for complete reconnection
Dam removal	Mainstem channels	Change reservoir to riverine environment; increase downstream flux of sediment, wood, water, and energy	Change from lentic to lotic system; increase biotic exchange in up and downstream direction	Infrastructure within the mainstem and floodplain may not allow for complete reconnection

the reconnection of two former meander bends that are now floodplain sloughs (Pess *et al.*, 1999). However, the river channel incised 1 to 2 m between the period when the bends where cut off in 1929 and 1991 making reconnection of the former meanders difficult (Collins *et al.*, 2003). This problem is common to many stream channels in Europe, North America, and throughout the world.

Submersible check dams are technique used to raise the bed elevation of rivers with incised mainstem channels and reconnect them with their former channels and floodplain. For example, in the Danube River in Slovakia, check dams in lateral artificial channels have been used to aggrade the river to the point where older channels are now reconnected, improving the retention time of water in the reach (Cowx and

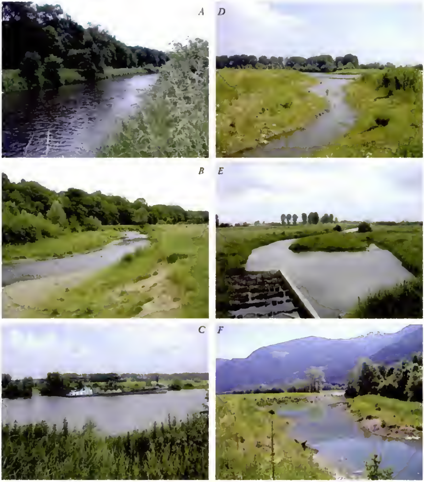
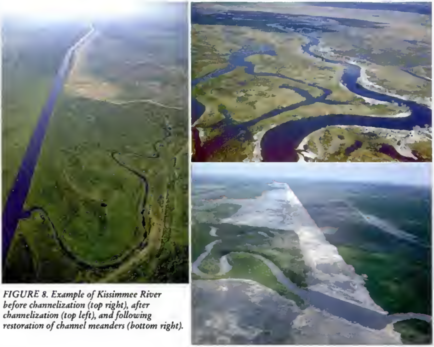


FIGURE 7. Photographs of some common floodplain rehabilitation techniques including bank protection and removal of bank protection (Photo A) straightened reach, Photo B downstream restored reach, River Aggar, Germany), active gravel mining operation (Photo C) and reclaimed gravel pit on the Rhine River, Germany (Photo D), reconnection of former oxbow or side-channel (Photo E, Rhine River, fishway in foreground), and excavation of new off-channel habitat (Photo F, River Drau, Austria).



SOUTH FLORIDA WATER MANAGEMENT DISTRICT

FIGURE 8. Example of Kissimmee River before channelization (top right), after channelization (top left), and following restoration of channel meanders (bottom right).

Welcomme, 1998). On the Kissimmee River, Florida (USA), channel filling has been used to reconnect old meanders and floodplains (Toth *et al.*, 1993; Figure 8). The main channel has been filled from levee material at points in the river where the meander crosses the main channel. This method only works if the floodplain or some of its key habitats remains intact (Cowx and Welcomme, 1998).

2.3.1.2 Levee breeching and setbacks

A levee breeching or setback project, whether full or partial, allows a river to migrate and subsequently create and maintain different floodplain channel types, thereby increasing the habitat diversity of a floodplain. The relative extent of such rehabilitation actions is often limited by land constraints and thus many such projects are a combination of full and partial levee removal. One technique is a "beaded approach" to floodplains, where small sectors of the full floodplain width (e.g. 6 to 8 km long) are allowed to function, alternating with leveed sections of the river. In these full floodplain width sections, several habitat features are included such as floodplain channels and wetlands. This "family" of habitats will allow the river to function properly and possibly create sediment, flow, and nutrient pulses (Sparks, 1995). This technique allows portions of the floodplain to be inundated and encourages scour, erosion, and deposition in those areas. Such a technique can be used in areas where natural channel migration is normally high such as the confluence with tributary junctions.

2.3.1.3 Channel reconstruction or meandering

Depending on the stream power and sediment load and composition, a channel may revert back to a meandering channel if left undisturbed for many decades

(Brookes, 1992). However, most floodplain channels tend to be low energy and require direct intervention to be restored to a meandering channel connected with its floodplain. Therefore, one of the more common techniques for rehabilitating rivers and reconnecting them to their floodplains is to construct a new meandering channel (Cowx and Welcomme, 1998; Pess *et al.*, 2005). This technique is particularly common in Denmark and northern Europe where most rivers are low gradient, highly channelized, and unlikely to return to their historical sinuous channel type without intervention (Brookes, 1992; Hansen, 1996). Restoring the sinuosity of a stream is done by either 1) pulling back levees and constructing a new meandering channel (rather than allowing the stream to naturally form its own channel as with a levee set back or removal project), 2) constructing a meandering channel adjacent to the straightened channel and then diverting the river into the new channel, or 3) if remnants of the original meandering channel exists adjacent to the straightened channel, diverting the straightened channel into its old channel, or 4) some variation of these three approaches. Many smaller channels have been straightened and piped and "daylighting" or exposing these concealed watercourses and then restoring natural remeandering pattern is an increasing common technique in urban and agricultural landscapes (Nielsen, 1996; Riley, 1998). Sometimes in highly incised channels it may not be feasible to aggrade the channel and reconnect it with the historical floodplain; a new floodplain is created by excavating the banks and widening the channels (Cowx and Welcomme, 1998). This is often coupled with a two-stage channel design, which incorporates a low flow channel and a new high-flow channel or new floodplain above the low flow channel but below the historic floodplain (Iversen *et al.*, 1993). All these methods seek to reconnect the river with its floodplain, increase the diversity of habitats (mixture of slow and fast water habitats), and lengthen the stream channel (reduce the gradient).

2.3.1.4 Construction of new floodplain habitats

The creation of new floodplain habitats is a form of habitat enhancement that involves active construction of new floodplain channel, ponds, or backwaters. Creation of new habitats can be an important form of enhancement, replacing natural floodplain channels that have been lost owing to flood control activities and other floodplain-isolating actions (Beechie *et al.*, 2001). There are two major habitat types that are typically created in river floodplains: side channels (surface and groundwater-fed) and backwaters or off-channel ponds (e.g. gravel pits, mill ponds, mine dredge ponds, and alcove ponds).

Creation of surface or groundwater-fed side channels are constructed to provide spawning habitat for adult fishes and rearing habitat for juveniles (Lister and Finnegan, 1997). They are typically connected at both the upstream and downstream ends to the main river channel, with an intake structure on the upstream end. Due to the surface water source, there is flexibility in choosing a project location. However, along with this water source comes sedimentation problems. Because of siltation concerns, this type of channel may be unsuitable for systems with high suspended sediment loads. Also, the location of the channel intake is critical for controlling sediment and organic introduction into the channel. Typically, a surface-fed side channel requires a dike or control structure and bank armouring at the upstream end to protect the channel from river flooding and bank erosion (Lister and Finnigan, 1997).

If the water table is low, groundwater-fed side channels can also be excavated. Groundwater-fed channels offer stable year-round water flows with little sediment, clean substrates and stable water temperatures (Sheng *et al.*, 1990; Bonnell, 1991). Groundwater channels are excavated parallel to the main river channel, often along an existing intermittent stream or relic channel. Many of these side channels were originally constructed as channels with nearly uniform depth and substrate to serve as spawning habitat for salmonids (Sheng *et al.*, 1990 from Giannico and Hinch; Giannico and Hinch, 2003).

Off-channel ponds can be created through several excavation or construction methods including excavation, blasting, damming or flow control structure on an existing perennial or intermittent stream, pond, or wetland (Cederholm *et al.*, 1988; Slaney and Zoldakas, 1997; Cowx and Welcomme, 1998). The construction of alcoves or backwaters directly connected and adjacent to the river is another technique that is popular in both North America and Europe (Slaney and Zoldokas, 1997; Cowx and Welcomme, 1998). Alcoves have been constructed through excavation of a pond directly adjacent to the river channel. Other forms of constructed habitats include the construction of new side channels (surface or groundwater fed), excavation of backwaters or alcoves adjacent to the main channel, and excavation of ponds and other wetlands. Regardless of how the ponds or channels are constructed, it is important to ensure that these constructed habitats are connected to the river as least seasonally, but preferably consistently throughout the year. However, floodplain ponds in developing countries such as Nigeria are sometimes purposefully excavated so that water and fish will be isolated when floodwaters recede—thus providing fishing opportunities (Neiland and Ladu, 1998).

Floodplain mining and other floodplain excavation activities have resulted in the creation of gravel pits, mill ponds, and mine-dredge ponds along river channels (Norman, 1998). The restructuring and connection of these ponds with the river channel provides opportunities for additional floodplain habitats. These areas can provide important rearing habitat for a variety of fishes that prefer slow water habitats, though in some areas they are thought to provide refuge from predators and exotic species (Bryant, 1988; Swales and Levings, 1989; Richards *et al.*, 1992; Norman, 1998; Hall *et al.*, 2000). The successful conversion of gravel pits, mill ponds, and other constructed floodplain lakes depends on several factors including depth, morphology, adequate prey resources, cover, water quality, and adequate water quality (Norman, 1998). Shallow ponds with complex shapes provide the best opportunities for successful use by juvenile fishes, as large and deep ponds will provide less prey, as well as substantially higher risk of avulsion (Peterson, 1982; Norman, 1998).

2.3.2 Effectiveness of floodplain rehabilitation techniques

2.3.2.1 Isolated habitats

Similar to other rehabilitation techniques, the effectiveness of floodplain rehabilitation can be measured through recovery of processes (sediment transport, hydrology, connectivity), and physical and biological response. The rehabilitation of connectivity and physical and biological response have been frequently examined, though the affects on watershed processes have infrequently been measured. In terms of restoring connectivity and improvements in physical habitat, the benefits are obvious and can be measured via the amount of physical habitat connected. For example, in the Danube River the reconnection of a former side channel and canal in resulted in an additional 50 km of habitat. Reconnected floodplain ponds has proven to be effective at providing habitat for juvenile salmonids such as coho (*Oncorhynchus kisutch*) and Chinook (*O. tshawytscha*) salmon (Richards *et al.*, 1992; Norman, 1998; Roni *et al.*, 2002) in western North America. They are also known to provide critical rearing habitat for a number of other fishes including cyprinids (minnows), catastomids (suckers), and many other warm and coolwater fishes. Schmutz *et al.*, (1994) reported the colonization of a reconnected section of the Danube River and found over 40 species of fish after only one year. In the lower Rhine River, reconnection of floodplain lakes and channels lead to an increase in rheophilic cyprinids (Grift *et al.*, 2001; Simons *et al.*, 2001). The level of flow and connection of habitats to the main Rhine was positively correlated with rheophilic cyprinid presence and abundance (Grift *et al.*, 2001). Similarly, the reconnection of a side channel/canal in the Danube allowed rapid colonization by many species, though pike-perch (*Stizostedion lucioperca*; an important sport and commercial fish) did not respond as favourably as other species due to their aversion to bypass channels around weirs (Schmutz *et al.*, 1998). Rapid colonization is not altogether surprising given that many freshwater fish species are entirely or partially dependent upon floodplain habitats for all or portions of their life history (Mann, 1996; Welcomme, 1985; Cowx and Welcomme, 1998; Buijse *et al.*, 2002).

In Bangladesh and India, reconnection of secondary channels and floodplain lakes has been shown to be an effective practice to increase fish catch (Thompson and Hossain, 1998; Rahman *et al.*, 1999). For example, the reconnection of Singharagi Beel lead to an increase in annual fisheries yield from 1 863 kg/ha to 11 384 kg/ha (partly because of increased effort) and the percentage of catch composed of large migratory catfish and carps increased from 2 to 24 percent (Payne and Cowan, 1998). While not often evaluated, reconnection of isolated habitats is believed to be one of the more effective techniques from both a physical and biological standpoint, partly because it relies on existing habitat rather than enhancing habitat or creating new habitat (Roni *et al.*, 2002; Pess *et al.*, 2005). Studies from various developed and developing countries support this hypothesis.

2.3.2.2 Levee breaching

Most of the information on project effectiveness for levee breaching and setbacks has focused on physical aspects and the lateral hydrologic connectivity of habitats. The removal of levees and bank armouring allow the channel to migrate naturally and recover its former sinuosity and is becoming increasingly common. Early results suggest these projects lead to successful improvements in floodplain processes such as nutrient transport and lateral erosion. Change in nutrient transport levels have been reported for levee removal projects that allow reconnection with wetlands (Childers *et al.*, 1999a,b). Florsheim and Mount (2002) demonstrated that levee breaching on the Cosumnes River, California, allowed for the successful restoration of floodplain features such as sand-splays complexes. Evidence from ongoing studies in Austria indicates that following levee removal river channels begin to move laterally and recover some sinuosity and habitat complexity fairly quickly (Jungwirth *et al.*, 2002; Muhar *et al.*, 2004). Levee set backs and modifications on the Danube River have also been shown to benefit not only rheophilic fishes, but also amphibians and dragonflies (Chovanec *et al.*, 2002). Moreover, Hein *et al.*, (1999) demonstrated increased plankton biomass in reconnected habitats and that plankton production declined as connectivity of habitat decreased. These and other Austrian studies have demonstrated not only improvements in physical habitat and restoration of natural erosional and channel migration processes, but also improvements in both fish and riparian diversity and age structure (Jungwirth *et al.*, 2002).

2.3.2.3 Remeandering

The effectiveness of remeandering can be measured in part by increase in the total stream length as the amount of total stream length can increase dramatically by reinstating meanders. In large rivers, the amount of length lost due to straightening can be dramatic. For example, it is estimated that channelization of the Mississippi River in the United States reduced stream length by 229 km (Gore and Shields, 1995). Hvidsten and Johnsen (1992) reported that channelization of a sinuous reach of a Norwegian stream reduced stream length from 7.5 to 2.5 km. The increase in stream length by remeandering can be equally impressive. For example, in a review of Danish stream rehabilitation, Iversen *et al.* (1993), reported increases in stream length ranging from 17 percent to more than 60 percent for five river meander reinstatement projects. Clearly restoring stream and floodplain meanders dramatically increases total available habitat and habitat diversity.

The remeandering on the Brede, Cole, and Skerne rivers in Denmark and England is one of the more thorough ongoing evaluations of stream remeandering. Evaluation of these projects indicated an obvious improvement in habitat complexity and channel morphology, flood frequency, and amount of water passing onto the floodplain (floodplain connectivity), as well as an increase in sediment deposition and sediment-associated phosphorous (Kronvang *et al.*, 1998; Sear *et al.*, 1998). Studies of macroinvertebrates, fish fauna, and aquatic vegetation in the River Gelså and other remeandered Danish streams have shown some positive results (Iversen *et al.*, 1993; Hansen, 1996; Friberg *et al.*, 1998). Improvements in both physical habitat and fish species diversity have also been reported from Austrian streams (Jungwirth *et al.*, 1995).

The positive benefits of restoring or recreating meanders on stream morphology and hydrology and sediment transport have been documented. Similarly, recovery of macroinvertebrates and other biota from construction impacts appears to be relatively rapid. However, long-term studies conclusively documenting recovery or increase in fish production have not yet been completed. As with many structural manipulations of physical habitat, if potential water quality problems are not addressed, they can impair biotic responses to floodplain habitat rehabilitation.

2.3.2.4 Constructed habitats

The connectivity of the habitats to the main river channel plays a large role in their physical effectiveness. If the habitats remain connected through a variety of flows they are much more likely to be used by a variety of fish life stages. Constructed floodplain habitats have been shown to be particularly effective at both providing habitat for juvenile salmonids and increasing their survival (Lister and Bengeyfield, 1998; Solazzi *et al.*, 2000; Giannico and Hinch, 2003). For instance Solazzi *et al.* (2000) concluded that creation of slow-water floodplain channel habitats increased overwinter survival for coho salmon, cutthroat trout (*Oncorhynchus clarki*), and steelhead (*O. mykiss*). Chovanec *et al.* (2002) reported rapid colonization by rheophilic fish species (e.g. *Chondrostoma nasus*) of constructed backwaters and side channels created on the Danube River in Austria. Similarly, Langler and Smith (2001) found age 0 coarse fish numbers and diversity higher in created backwater habitats than unmanipulated habitats in the Huntspill River, England. Habersack and Nachtnebel (1995) found that a constructed side channel of the River Drau, Austria, had higher diversity of habitats, substrates, macroinvertebrates and higher densities of fish in the side channel than in other river reaches. Excavation of groundwater channels is a particularly popular technique for creating spawning habitat for salmonid fishes (Bonnell, 1991; Cowan, 1991; Hall *et al.*, 2000). Similar to surface-fed side channels, groundwater-fed channels also provide rearing habitat for juvenile fishes, particularly coho salmon (Bryant, 1988; Bonnell, 1991; Richards *et al.*, 1992). These studies demonstrate that properly constructed floodplain habitats can provide important spawning and rearing areas for fishes.

2.3.3 Conclusions – floodplain habitats

Floodplain rehabilitation is a relatively new science and long-term studies documenting biological effectiveness are not currently available. In addition, the goals are typically broad ecological and cultural objectives. Thus trying to evaluate a purely fisheries response to a project is difficult. It is clear that most techniques described can lead to improvements in physical and hydrologic and other natural processes, provide additional slow water habitats, and additional habitat for fishes. Below we summarize some of the key findings and concerns.

- Reconnection of isolated floodplain habitats is particularly effective at improving habitat diversity, providing habitats for various fishes, and increasing species diversity.
- Levee removal, channel remeandering, and construction of floodplain habitats have all shown promising results both physically and for biota, but long term data on their success are not yet available
- Many factors influence the physical and biological effectiveness of projects including: habitat complexity, depth, wood volume, connectivity with the main channel, channel incision, flow volume and source, exotic species, project size, and upstream flow regulation and water quality.

2.4 DAM REMOVAL AND FLOOD RESTORATION

2.4.1 Techniques for dam removal and flood restoration

Dam removal, weir removal, creation of fish passage or bypass channels, and restoration of peak flows in regulated rivers are employed primarily to restore longitudinal connectivity of stream and floodplain habitats, improve fish access, allow for natural transport of water, sediment, organic material and nutrients, and maintain or restore natural riverine processes that create and maintain fish habitat (Pess *et al.*, 2005). While construction of dams and weirs is still occurring at a rapid pace in developing countries, many dams and diversion weirs in developed countries have reached the end of their useful life, and are being considered for removal or restructuring because of ecological concerns, as well as the safety concerns and costs of continued operation or repair of older structures (Stanley and Doyle, 2003). For example, in the last 20 years over 500 dams have been removed in the United States, mostly smaller dams less than 20 meters high with a storage area of less than 100 acre-ft (123 000 m^3) (Poff and Hart, 2002; Stanley and Doyle 2003). Similarly, removal of weirs and low-head dams or installation of bypass channels and fish passage structures is one of the most common techniques for improving fish passage in Europe (Iversen *et al.*, 1993; Cowx and Welcomme, 1998; DVWK, 2002) and other parts of the world (Larinier, 2001; Larinier and Marmulla, 2004).

Dams negatively impact aquatic resources by blocking migration of fishes, dramatically impacting watershed processes by changing sediment, nutrient, hydrologic and temperature regime downstream of the dam and by converting areas above the dam from a lotic to a lentic environment (Pejchar and Warner, 2001; Jackson and Marmulla, 2001; Pizzuto, 2002; Poff and Hart, 2002; Arthington *et al.*, 2004). Most studies of dams in other regions have focused on the impacts of dams rather than removal of dams. Similar to dam removal studies, these studies demonstrate changes in physical conditions or biota (e.g. Grzybkowska *et al.*, 1990; World Commission on Dams, 2000). The consideration of these and other negative ecological consequences of dams on aquatic resources, particularly migratory fish, have made the removal of both small and large dams an increasingly common method of rehabilitation (Chisholm, 1999).

Similar to dam construction, dam removal affects three areas—upstream of a dam reservoir, a reservoir/impoundment area, and below a dam. Biotic exchange between a river system below and above a dam increases after dam removal, as does the role of aquatic migratory species (e.g. salmonids) that may have been blocked because of a dam (Hart and Poff, 2002). The former reservoir or impoundment will experience dramatic changes in aquatic and riparian biota, flow characteristics, sediment erosion and deposition, and the channel is likely to be unstable in the first few years after dam removal. Dam removal typically increases downstream sediment supplies and changes flow patterns and result in significant changes in instream and riparian biota (Bushaw-Newton *et al.*, 2002; Nislow *et al.*, 2002; Shafroth *et al.*, 2002). This suggests that the riparian environment will change dramatically as floods remove or change vegetation composition and create habitat suitable for more flood-tolerant species. Increases in sediment can generally be expected to raise bed elevation and increase lateral movement where there is room within the floodplain (Pizzuto, 2002; Shafroth *et al.*, 2002). Such changes should result in more depositional bars and other newly created landforms that would favour early successional riparian species as well as create a diversity of aquatic habitats.

Another approach to restoring a natural hydrologic regime when dam removal is not feasible is the restoration of natural flood or flow regime. The regulation of river flows has lead to changes in sediment, habitat connectivity, biotic diversity, colonization of riparian plants, nutrient cycling and reduction in primary productivity (Osmundson *et al.*, 2002). Many riparian species such as cottonwood become established under specific flow conditions, such as floods (Fenner *et al.*, 1985; Auble and Scott, 1998). When floods are eliminated, the natural establishment of such species becomes problematic. Additionally, species that would not become established because of the scouring of floods appear in places they are not regularly seen. Flooding is also important for colonization of constructed riverine habitats by fishes. For example, in a constructed riverine wetland in the upper Mississippi River, fish species diversity increased dramatically following flooding (Theiling *et al.*, 1999). Thus the timed release of waters from dams to mimic natural flows at certain times is becoming recognized as a valuable rehabilitation technique (Rood and Mahoney, 1993).

2.4.2 Effectiveness of dam removal and flood restoration

2.4.2.1 Effectiveness dam removal

Because dam removal is relatively new technique, there is not extensive published literature on its effectiveness, though many of the benefits are inherently obvious (e.g. fish access and passage). Below we summarize the results of studies that have examined the effects of dam and weir removal on connectivity, processes, habitat and biota below the dam, and processes, physical habitat, and biota immediately above the dam.

The restoration of upstream-downstream connectivity and fish access is one of the most clearly demonstrated effects of dam removal. Hart *et al.* (2002) summarized the results of several dam removal projects in the United States and reported more than 10 cases of dam removal that resulted in rapid colonization of former impoundment sites and upstream areas by both migratory and resident fishes in both warm and coolwater rivers. For example, dam removal on the Clearwater River, Idaho (USA) in 1963 reconnected the main stem, increasing both habitat quality and Chinook salmon runs (Shuman, 1995). Similarly, removal of 150-year-old Edwards Dam on the Kennebec River in Maine (USA) resulted in large numbers of American eel (*Anguilla rostrata*), alewife (*Alosa pseudharengus*), and Atlantic and shortnose sturgeon (*Acipenser* spp.) moving upstream within the first year, as well as juvenile downstream migrants in subsequent years (Hart *et al.*, 2002). Smith *et al.*, (2000) reported improved fish passage following removal of a 3 m high dam on a stream in Oregon (USA), but continued water withdrawal and other factors upstream of the dam prevented full recovery of both physical and biological conditions. Kanehl *et al.*, (1997) also examined the effects of removal of a low head dam in the Wisconsin (USA) and found improvement in habitat quality, biotic integrity, and smallmouth bass (*Micropterus dolomieu*) abundance and biomass five years after dam removal. While not discussed extensively in this report, the installation of bypass channels to allow fish access above diversion weirs has also proven to be highly successful rehabilitation technique for fishes in European streams (DVWK, 2002; Iversen *et al.*, 1993). Clearly dam removal has a number of benefits for migratory and lotic fishes.

Below a dam several major changes occur, the most obvious being a change in sediment and channel form due to a change in sediment flux. Several studies have demonstrated changes in sediment transport and fine sediment, but the changes in sediment depend upon the composition and levels of fine sediment trapped behind the former dam. For example, Doyle *et al.* (2003) examined low-head dam (<3 m high) removal in two Wisconsin rivers and erosion of fine sediment deposited in the former reservoir and increased deposition of fine sediment downstream. Hart *et al.* (2002) reviewed 20 dam removals in the US, 14 of which documented increased sediment transport, but few studies were long enough to document changes in the channel downstream of the dam. Other changes include a return to a more natural temperature regime, plant colonization, and a greater exchange of nutrients and organic matter with upstream portions of a watershed (Hart *et al.*, 2002). Downstream effects from dam removal on ecological attributes ultimately depend on how reservoir-derived deposits move into and through downstream reaches (Stanley and Doyle, 2003). Changes in downstream water temperatures also typically occur following dam removal as do shifts in the macroinvertebrate community (Hart *et al.*, 2002). Whether temperature increases or decreases following dam removal depends on several factors including the previous operations and structure of the dam and reservoir.

Chisholm (1999) reported anecdotal evidence on the positive effects of 25 dam removals in the United States. However, he also provided information on one dam removal (Fort Edward Dam on the Hudson River in New York State) that is considered a failure because it released heavy metals and pollutants trapped in reservoir sediments, which continue to have negative consequences on aquatic resources downstream from the former dam site. In addition, increase in turbidity and sediment immediately following and shortly after dam removal are not uncommon.

Former impoundments are affected by dam removal because they are returned to river, riparian, and floodplain habitats (Stanley and Doyle, 2003). This physical change reduces the residence time of water in a former reservoir reach, and subsequently reduces the amount of sediment and other materials stored within a reach. This in turn shifts the biota from a lentic to a lotic system (Hart and Poff, 2002). For example, fish and macroinvertebrates adapted to a high sediment supply reservoir environment gave way to riverine fish and macroinvertebrates within a year of two separate dam removal projects in Wisconsin (Stanley *et al.*, 2002; Stanley and Doyle, 2003). In a related study on the Baraboo River, Wisconsin, Stanley *et al.* (2002) found that within 1 year macroinvertebrate assemblages above in formerly impounded stream reaches (reservoir) were similar to those in upstream and downstream reaches. Clearly dam removal will also result in changes in algae and other aquatic vegetation as well as riparian vegetation in the former reservoir sites (Hart *et al.*, 2002; Shafroth *et al.*, 2002). Shafroth *et al.* (2002) found that plants that colonized newly exposed soil along former reservoir bottoms were often composed of hydrophilic, monotypic species such as stinging nettle (*Urtica dioica*) and red canary grass (*Phalaris arundinacea*), over time sites became drier and dominated by xeric species. These studies demonstrate the dramatic changes in physical habitat and riparian and aquatic flora and fauna that occur following dam removal. They also suggest that there are some negative consequences of dam removal such as short-term channel instability or colonization of newly exposed riparian areas by invasive riparian species.

2.4.2.2 Effectiveness of flood restoration

Because restoration of flood flows is a relatively new technique, there is somewhat limited information on its effectiveness, though the results have been very positive. The most notable test case has been the attempts to restore flood flows in the Grand Canyon in the United States. Along the Colorado River in the Glen Marble and Grand Canyon reaches, forests and marshlands have become established in areas that were formerly scoured by floods (Stromberg, 2001). Results of high flow tests have show some promising results in this large river system that formerly had a very dynamic flow regime prior to regulation. Recent restoration activities that simulate floods have resulted in changes to the colonization of riparian areas that more closely mimic natural colonization patterns and reduced nearshore woody vegetation that artificially established as a result of flood control (Mahoney and Rood, 1990; Ellis *et al.*, 2001; Stevens *et al.*, 2001). Stevens *et al.*, (2001) monitored the effects of flood simulations downstream of Glen Canyon Dam on the Colorado River and found that floods restored sandbar habitat and inundated patches of woody vegetation with as much as 1 m of sand. However, the woody vegetation was not eliminated and backwater marshlands that had established after dam construction also remained. Similarly, Hill and Platts (1998) monitored the effect of altered flows in the form of flood simulations and increased base flows in the Owens River, California. They found that riparian vegetation rapidly recovered following the changed flow regime and that instream habitat and fish abundance substantially improved concurrent with the changes in vegetation. Speierl *et al.* (2002) reported an increased in both species diversity and abundance following restoration of natural flow dynamics in the Main and Rodach rivers in Germany. Other studies have indicated that survival and growth of seedlings of some riparian plant species is higher under natural versus regulated flow conditions (Johansson and Nilsson, 2002). Similarly, restoration of instream flows can lead to recovery of riparian forests, birds, and fish populations (Rood *et al.*, 2003). Studies on natural floods suggest the importance of these events for riverine ecosystems and support the use of flood restoration in regulated rivers. Large floods in the 1980s and 1990s on the Missouri River reconnected many wetlands and floodplain lakes isolated through decades of river regulation and channelization (Galat *et al.*, 1998). Post flood studies indicated that reconnected areas had a more diverse fish fauna than areas that remained isolated (Galat *et al.*, 1998). Similarly, high water years lead to reconnection of floodplain wetlands in the Green River, Utah (USA), and provided rearing habitat for endangered larval razorback sucker (*Xyrauchen texanus)* not available in the main river channel (Modde *et al.*, 2001). These studies on natural floods and initial studies from restoration of flood flows suggest positive benefits for sediment transport, riparian vegetation, and aquatic biota.

Similar to increases in high flows, adequate instream flows or base flows, are needed to maintain aquatic and riparian habitat and production of aquatic ecosystems and biota (Petts and Maddock, 1996; Stanford *et al.*, 1996; Annear *et al.*, 2002; Arthington and Pusey, 2003). Increasing instream flows on rivers with water diversions or rewatering stream reaches are methods known to reduce water temperatures, improve water quality, and generally benefit biota (Weisberg and Burton, 1993; Petts and Maddock, 1996). There is an extensive body of literature on instream flows and it would be difficult to treat them adequately in a review of physical habitat rehabilitation techniques. However, along with restoring floods it is important that minimum instream flows are established to protect biota and assure the success of other habitat rehabilitation actions.

2.4.3 Conclusions – dam removal and flood restoration

Initial studies on dam removal and restoration of flood flows and a natural hydrology regime show promising early results at restoring access, natural processes, habitats, and recovering lotic biota. Based on the review of existing studies we provide the following recommendations:

- Dam or weir removal appears to be highly effective at restoring processes, though long-term monitoring of recovery is lacking.
- Fish and other migratory biota rapidly colonize areas upstream of dams.
- Some negative impacts such as fine sediment or water quality may be apparent depending on the level and contents of sediments stored in the reservoir and whether attempts are made to remove or stabilize them.

- In the absence of dam removal, restoration of flood flows has shown promising results in terms of restoration processes, reconnecting habitats, and restoring flood-dependent biota.

2.5 INSTREAM HABITAT STRUCTURES

2.5.1 Common instream habitat structures

The placement of physical structures into lotic environments to create pools, to alter channel morphology, and to provide cover and habitat for fish and other aquatic organisms has a long history (White, 1996, 2002). It was one of the first methods used to mitigate habitat degradation and to increase fish production in streams and rivers (Tarzwell, 1934, 1937, 1938) and also is one of the most common and widespread methods in use. Many different configurations of instream structures and methods have been used over the years to improve habitat (White and Brynildson, 1967; Vetrano, 1988; Hunt, 1993; Riley and Fausch, 1995; Cowx and Welcomme, 1998). These techniques involve placement of materials such as large woody debris (LWD), boulders, and other materials into the active stream channel or actual manipulation of the active channel itself in an effort to improve fish habitat. They can be categorized by purpose (e.g. create pools, trap gravel) and material and can include such structures as boulder or log weirs, dams and deflectors, cover structures, rootwads and brush bundles, gabions, logjams and placement of spawning gravel (Table 8; Figure 9). Because these activities seek to enhance habitat rather than restore a deficient process (e.g. riparian, hydrology) or return a stream to some predisturbance state, they are called instream habitat enhancement rather than restoration.

TABLE 8

Common types of structures placed in streams to improve fish habitat. Adding channel meanders is discussed in section on floodplain rehabilitation. Modified from Roni (2005).

Type of instream rehabilitation	Definition	Typical Purpose
Log structures (e.g. weirs, sills, deflectors, logs, wing deflectors)	Placement of logs or log structures into active channel	Create pools and cover for fish, trap gravel, confine channel, or create spawning habitat
Log jams (multiple log structures, engineered log jams)	Multiple logs placed in active channel to form a debris dam and trap gravel	Create pools and holding and rearing areas for fish, trap sediment, prevent channel migration, restore floodplain and side channels
Cover structures (lunker structures, rock or log shelters)	Structures embedded in stream bank	Provide fish cover and prevent erosion
Boulders structures (weirs, clusters, deflectors)	Single or multiple boulders placed in wetted channel	Create pools and cover for fish, trap gravel, confine channel, or create spawning habitat
Gabions	Wire mesh baskets filled with gravel and cobble	Trap gravel and create pools or spawning habitat
Brush bundles/rootwads	Placement of woody material in pools or slow water areas	Provide cover for juvenile and adult fish, refuge from high flows, substrate for macroinvertebrates
Gravel additions and spawning pads	Addition of gravels or creation of riffles	Provide spawning habitat for fishes
Rubble matts or boulder additions to create riffles	Addition of boulders and cobble to create riffles	Increase riffle diversity (velocity and depth), create shallow water habitat
Sediment traps	Excavation of a depression or pond in active channel to trap fine sediment	Improve channel conditions and morphology, and increase grain size
Channel reconstruction and realignment (see floodplain rehabilitation section)	Alter channel morphology by excavating new channel to restore meander patterns or return to historic channel	Restore meander patterns, increase habitat complexity and pool-riffle ratio, reduce channel width

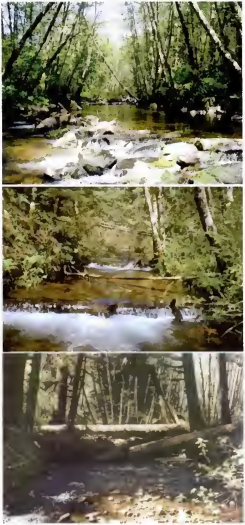


FIGURE 9. Three common instream habitat enhancement techniques or structures: rock weir (top), log weir (middle), and log jam or multiple log structure (bottom).

Most instream enhancement techniques used today were developed in the 1930s for use in the relatively low-gradient streams of the midwestern United States (Hunt, 1993; White, 1996). They were designed to enhance trout habitat that had been simplified and degraded through agriculture, forestry, and other land use. The primary purpose of early instream structures was to narrow channels, provide cover for trout and other game fish, and reduce mortality due to predation (Tabor and Wurtsbaugh, 1991; Gregory and Levings 1996). Instream enhancement techniques developed in the Midwest were later applied to streams throughout North America and Europe with varying degrees of success (Ehlers, 1956; Armantrout, 1991; Iversen *et al.*, 1993; O'Grady, 1995; Cowx and Welcomme, 1998; Gortz, 1998; Laasonen *et al.*, 1998). In higher energy streams, such as those in mountainous areas of western North America, the major goal of placement of instream structures has been to create pools and trap gravel. In more recent efforts in low energy, low gradient (< 1 percent gradient) channelized streams such as those found in floodplains and agricultural land, the goals are generally to increase habitat heterogeneity (create a suite of both shallow and deep water habitats), and either narrow or widen the channel depending on the channel and its level of degradation.

2.5.2 Effectiveness of instream structures

Though instream habitat enhancement is popular, widespread, has a lengthy history, and more than 60 published evaluations exist, most techniques have not been completely evaluated. The need for rigorous monitoring and evaluation has been noted for many decades (Tarzwell, 1937; Reeves *et al.*, 1991; Roni, 2005). There is considerable debate about the appropriateness of instream enhancement, because these techniques historically occurred without an assessment of what factors caused the lack of habitat complexity, what processes in the watershed might be disrupted and need to be corrected, and what factors might be limiting the physical and biological production of a system (Frissell and Nawa, 1992; Roni *et al.*, 2002; White, 2002). For example, the failure of instream wood and boulder structures to increase fish abundance in a stream reach may be due to a lack of riparian vegetation, caused in turn by land use activities such as agriculture or even urbanization. In such cases, if the underlying cause of the problem is not addressed, instream enhancement may represent only a short-term improvement in habitat and in heavily modified areas, it may be the only feasible option. Thus habitat enhancement projects are not without merit. Short-term habitat improvements may be needed to protect rare fishes and biota or to provide short-term benefits where some watershed process cannot be restored (e.g. woody debris in urban streams). They are particularly relevant when coupled with restoration of natural processes, such as riparian rehabilitation or sediment reduction (i.e. road treatments and slope stabilization), otherwise these habitat enhancement techniques run the risk of treating the symptoms rather than addressing the underlying problem causing the degradation or simplification of instream habitat. It is important to reiterate that instream habitat enhancement techniques seek to improve habitat rather than restore a disrupted process. Below we review their physical and biological effectiveness.

2.5.2.1 Physical habitat

Durability—The majority of monitoring and evaluation efforts of instream projects have focused on the effects of the project on channel morphology and instream habitat (Hunt, 1988; Reeves *et al.*, 1991; Binns, 1999; Roni *et al.*, 2002). Early evaluations

TABLE 9

Summary of key studies evaluating instream structure durability. Note that most studies did not break down success rate by structure type. NA = not available and a blank space indicates that that type of structure was not examined. Modified from Roni *et al.* (2002).

			Percent of structures functioning					
Study	N	Years after placement	Log Weirs	Rock Weirs	Deflec-tors	Natural Logs or Jams	Gabions	Total
Ehlers (1956)a	41	18	33%	0%	50%	NA	0%	24%
House *et al.* (1989)a	812	1-8	NA	NA	NA	NA	NA	86%
Armantrout (1991)b	362	5	70%	93%	NA	88%	90%	85%
Frisseli and Nawa (1992)c	155	1-5	32%	46%	33%	74%		39%
Crispin *et al.* (1993)a	200	1-4	NA	NA	NA	NA	NA	98%
House (1996)a	22	6-12	100%	100%	100%	100%	100	100%
Thom (1997)e	143	1	NA	NA	NA	86%		86%
Roper *et al.* (1998)d	3946	NA	NA		NA	NA		84%
Schmetterling and Pierce (1999)	66	1	86	78	79	100		85%
Thompson (2002)b	27	40-60	100%	NA	18%	NA	NA	48%

a Functioning defined as in-place and functioning as intended
b Functioning defined as improving habitat
c Functioning defined as functioning as intended
d Functioning defined as in-place or largely in-place but shifted
e Functioning defined as no movement or moved less than one bankful width

reported on the durability of instream structures, whether they created pools, and their longevity (Ehlers, 1956; Gard, 1972; Armantrout, 1991; Frissell and Nawa, 1992). Reported failure rates for various types of wood and boulder structures in North American streams are highly variable, ranging from 0 to 100 percent (Roni *et al.*, 2002; Table 9). A few studies such as Ehlers (1956) and Thompson (2002) have examined long-term durability (>18 years) of habitat enhancement structures. For example, Thompson (2002) found that weirs, deflectors, and cover structures constructed in the 1930s and 1950s in two Connecticut rivers were no longer functioning. He also reported that failed cover structures have actually lead to reduced streamside vegetation and overhead cover and that some bank protection and grade control structures have prevented the recovery of the channel and increased erosion. This and other studies on functionality of instream structures demonstrate the need to thoroughly understand channel dynamics and long-term impacts of stationary structures.

The physical success of many structures varies widely and is influenced by many factors including structure type, materials and design, stream power, and often by the investigators definition of "failure" or "success". As previously mentioned, early failures of structures often resulted from the use of inappropriate structures or lack of understanding of larger watershed processes (e.g. Kondolf *et al.*, 1996). More recently, highly artificial techniques such as log weirs are being replaced with techniques that mimic natural wood accumulations or habitat. These newer methods have been demonstrated to be more durable in some instances and more effective at producing changes in habitat than traditional structures (Cederholm *et al.*, 1997; Thom, 1997; Roni and Quinn, 2001a).

Changes in habitat—Despite highly variable results in durability, many studies have reported large (>50 percent) and significant increases in pool frequency, pool depth, woody debris, habitat heterogeneity, complexity, spawning gravel, and sediment retention following placement of instream structures (e.g. Crispin *et al.*, 1993; Cederholm *et al.*, 1997; Reeves *et al.*, 1997; Binns, 1999; Roni and Quinn, 2001a; Bates *et al.*, 1997; Gerhard and Reich, 2000; Brooks *et al.*, 2004). Similarly projects in the midwestern United States have also demonstrated physical habitat changes including increased depth, cover, and narrower channels as a result of instream enhancement projects (Hunt, 1988; Kern, 1992; Avery, 2004). Other studies designed to aggrade highly incised stream channels have produced increases in depth, width, pool area, and bed elevation (reduced incision) following boulder weir placement (Newbury and Gaboury, 1988; Shields *et al.*, 1993; Shields *et al.*, 1995a).

Several European studies in low gradient channelized streams have demonstrated changes in physical habitat following placement of structures. Gerhard and Reich (2000) demonstrated increased habitat complexity, depth, organic matter retention, and macroinvertebrate species richness and abundance. Näslund (1989) reported increased depth and pool area in reaches of a Swedish stream treated with boulder weirs, while log and boulder deflectors decreased wetted area and boulder clusters had little effect on these parameters. Pretty *et al.* (2003) found increased depth and flow heterogeneity in rehabilitated stream reaches than control reaches in 13 English streams. Zika and Peter (2002) found increased pool volume and number following placement of LWD in a Swiss stream. Jungwirth *et al.* (1995) reported changes in physical habitat complexity (e.g. depth, velocity, substrate) following placement of instream structures in Austrian streams. These studies indicate that while a variety of factors can affect the level of response, many instream structures lead to substantial changes and improvements in physical habitats.

2.5.2.2 Biological effectiveness of instream structures

Biological evaluations of different enhancement techniques have been less frequently than physical evaluations and have produced inconsistent results depending on the technique, region, species and life stage examined, as well as the duration of monitoring. The majority of these evaluations have focused on trout or juvenile anadromous salmonids, which historically were the focus of most instream enhancement efforts in North America and Europe. Monitoring of instream enhancement has been more intensive in North America than most regions of the world. For some regions of the United States, there are compendiums summarizing the effectiveness of most instream rehabilitation projects within a particular region (e.g. Hunt, 1988; Binns, 1999; Avery, 2004). The results of these and other studies, however, have been variable and there is still much debate in the scientific community about the biological effectiveness of different instream enhancement methods. In an effort to summarize the extensive biological evaluations, the following subsections discuss juvenile and salmonid fishes (anadromous and resident), nonsalmonids, macroinvertebrates, and other biota.

Juvenile anadromous salmonids responses–In a synthesis of the effectiveness of such efforts for Pacific salmon (*Oncorhynchus* spp.), Roni *et al.* (2002) reviewed published evaluations of anadromous salmonid response to instream enhancement structures. They found that while many reported positive responses of juvenile coho salmon, steelhead, and cutthroat trout to instream enhancement, few of these responses were of long enough duration to detect statistically significant increases. Their synthesis and previous reviews (Reeves *et al.,* 1991; Beschta *et al.,* 1994; Chapman, 1996) emphasize the need for rigorous long-term evaluations of the responses of juvenile fishes to instream enhancement. While the results from these projects and those reviewed by Roni *et al.* (2002) are variable, they do indicate that when implemented correctly instream rehabilitation techniques benefit species and life stages of Pacific salmonids that prefer pool habitats.

Several studies have also demonstrated the positive effects of instream structure placement on Atlantic salmon. O'Grady (1995) reported higher levels of age 1+ Atlantic salmon and brown trout parr in sections of Irish streams treated with boulder structures including weirs, stone mats, and boulder clusters. Kelly and Bracken (1998) reported increased juvenile Atlantic salmon densities following placement of boulders, riprap, and deflectors in the Rye Water in Ireland, but no response for brown trout. In a similar study examining Atlantic salmon and brown trout in 13 Irish streams, Gargan *et al.,* (2002) found significantly higher levels of both Atlantic salmon and brown trout parr in treated stream reaches, but no difference for older trout or for trout or salmon fry. Clarke and Scruton (2002) reported use of gravel additions by Atlantic salmon and high numbers of parr at gravel addition sites, though their monitoring was discontinued after 2 years. These studies suggest that placement of instream structures benefit juvenile Atlantic salmon, a species with habitat preferences similar to steelhead (rainbow trout).

Resident salmonids response—Early work examining the effectiveness of instream structures on resident trout suggested moderate increases of abundance, growth, condition, and survival of trout species associated with placement of structures (Tarzwell, 1938; Gard, 1961). Three compendiums on the response of trout to enhancement of Rocky Mountain and midwestern United States streams summarize much of this work (Hunt, 1988; Binns, 1999; Avery, 2004). In a review of 71 different instream enhancement projects installed between 1953 and 1998 in Wyoming, Binns (1999) detected more than a two fold increase in wild trout abundance either after treatment or between treatment and reference reaches for the 46 projects where fish data were available. However, few evaluations included more than a few years of data collection.

Numerous techniques for improving habitat for trout including brush bundles, current deflectors, bank cover, cut or cover logs, and LWD placement have been evaluated in Midwest streams. Hunt (1988) and Avery (2004) examined evaluations of instream habitat enhancement efforts in a total of 103 Wisconsin streams. Streambank debrushing and installation of brush bundles produced rather disappointing trout response in 11 different streams in Wisconsin (Hunt, 1988; Avery, 2004). Conversely, bank cover and current deflector structures demonstrated excellent results in increasing trout mean size and biomass, with 75 percent of the projects examined showing an increase in local abundance of 25 percent or more (Hunt, 1988; Avery, 2004).

Other North American studies not covered in these reviews have produced various results. An early 2-year study found increases in number and biomass of brook (*Salvelinus fontinalis*), brown, and rainbow trout following addition of brush bundles, and a decrease following removal of overhanging brush in other stream reaches (Broussu, 1954). Quinn and Kwak (2000) found more rainbow trout in a rehabilitated reach of an Arkansas River following placement of instream structures and bank protection. Saunders and Smith (1962) found increased levels of brook trout 1 year after placement of combinations of weirs, deflectors, and cover structures in a small stream on Prince Edward Island, Canada, though no increase in fish growth was detected. Whole or longitudinally cut half logs, anchored mid-channel, are used extensively in the Midwestern and the Eastern United States to provide cover for trout in both streams and lakes (Hunt, 1993). However, in a Pennsylvania study, Hartzler (1983) found no effect of half log cover structures on harvest, abundance, or biomass of brown trout. The results of these North American studies on instream enhancement demonstrate that overall instream enhancement efforts may increase local trout abundance and condition. However, multiple types of enhancement practices were implemented in many stream reaches and few projects were monitored long term (>5 years), making determination of which method led to increased fish abundance unclear in most instances.

Rehabilitation on channelized European trout streams have produced more consistently positive than results from American streams. Näslund (1989) examined four types of habitat improvement structures in relatively steep (3.3 percent) gradient Swedish stream and found brown trout densities increased up to three fold compared with nearby reference reaches. The most successful structures were boulder weirs (dams) and log deflectors, while boulder clusters and boulder deflectors appeared to be ineffective. Linlokken (1997) monitored the response of brown trout before and after construction of boulder weirs (10 years total) and found a 3 fold increase in brown trout abundance. Hvidsten and Johnsen (1992) found higher levels of juvenile brown trout following placement of boulder weirs in a recently channelized section of the River Soya, Norway. Zika and Peter (2002) reported increased rainbow trout and brown trout following placement of LWD in a Swiss stream. O'Grady *et al.* (2002) and Gargan *et al.,* (2002) examined the effects of a variety of boulder and wood structures on brown trout in treatment and control reaches of 20 Irish streams and found higher levels of trout parr in treated stream reaches. Kelly and Bracken (1998) reported increased salmon densities but not brown trout in preliminary studies of boulder placement in the Rye Water, Ireland. The relatively consistent results of the European studies may be in part because the studies were more rigorous monitored than many North American studies, but also because the streams were highly simplified prior to enhancement and addition of instream structure may have resulted in larger changes in habitat quality (Roni and Quinn, 2001a).

Adult salmonids response—Where spawning gravels are in low abundance or of low quality, habitat structures such as channel-spanning LWD, boulder clusters, or gabions may recruit and store gravel (House *et al.,* 1989; House, 1996). The evaluations of adult salmonid response to enhancement structures or gravel placement have been limited to a handful of short-term studies that demonstrated adult salmonid use of gravel accumulated at structures (Avery, 1996; House, 1996; Gortz, 1998), or observations of redds or adult spawning near enhancement sites (Moreau, 1984; Crispin *et al.,* 1993; House, 1996; Iversen *et al.,* 1993) (Table 10). For example, Gortz (1998) found three

Table 10

Summary of studies reviewed that examined adult response to instream habitat improvement.

Study	Techniques	Observations
Crispin et *al.*,1993	Various wood and boulder weirs and structures throughout entire anadromous reach	Coho spawner abundance and redds throughout stream increased four fold in years following treatment compared pretreatment (11 years of monitoring)
House, 1984	Gabions and rock weirs	Increased steelhead and coho spawner abundance, and percentage of entire spawner population using treated stream reaches (3 years of monitoring)
House and Boehne, 1985; House, 1996	Gabion weir placement	2.5 fold increase in coho spawner abundance (12 years of monitoring)
Gortz, 1998	Gravel, boulders, and stream deflectors	Five fold in restored versus control reaches (1 year of monitoring)
Moreau, 1984	Boulder weirs, deflectors, and clusters	Documented steelhead and Chinook use of gravels trapped by placed structures. (No preproject data, 1 year of monitoring)
West, 1984	Gravel cleaning, boulder placement	Chinook salmon spawner use increased up to three fold in some areas (two years of monitoring)
Clarke and Scruton, 2002	Gravel additions	Initial results showed spawner use of gravels but monitoring was discontinued before conclusive results were obtained (2 years of monitoring)
Avery, 1996	Gravel riffles and sediment traps	No change in age 0 brook or brown trout (sediment traps rapidly filled with sand, 7 years of monitoring)
Klassen and Northcote, 1988	Gabions weirs	No difference in pink salmon (*Oncorhynchus gorbuscha*) egg to fry survival between gabions and natural sites (1 year of data)

times as many brown trout redds in restore reaches of the River Esrom, Denmark, but found low numbers of fry, suggesting the spawning was not successful. Vehanen *et al.* (2003) found that adult grayling preferred a section of a Finish stream enhanced with boulder and cobble structures (reefs and islands). Iversen *et al.* (1993) reported increased fry and trout numbers in rivers Gudenå and River Hjortvad in Denmark, though they suggested projects on other Danish streams were not as successful because gravel was placed in slow water areas or because of siltation or poor water quality. In some cases the creation of spawning riffles is coupled with sediment traps (e.g. Avery, 1996), which has shown some level of success, but it is unclear how long the sediment traps will function before needing to be reexcavated.

This lack of rigorous evaluation of adult salmon and trout response to instream enhancement stems in part from the multiple generations and thus long time frame (>10 years) needed to detect an adult response to habitat alterations (Bisson *et al.*, 1997; Korman and Higgins 1997). Nonetheless, evaluating adult response is critical for projects focusing on enhancement of spawning habitat and review of the limited published studies suggests increases in adult use of treated areas and increased fry (Table 10).

Nonsalmonid fish response—Less information exists on the response of nonsalmonid fishes to instream habitat enhancement (Bilby *et al.*, 1996; Lonzarich and Quinn, 1995; Roni 2003) (Table 11). A review of habitat enhancement projects in warm-water streams throughout the Midwest found that a wide variety of techniques were successful in altering stream morphology and increasing stream cover, although rigorous physical and biological evaluation of individual projects was generally lacking (Lyons and Courtney, 1990). Lonzarich and Quinn (1995) found no effect of woody debris cover and depth on threespine stickleback (*Gasterosteus aculeatus*) or coast range sculpin (*C. aleuticus*) habitat use, growth, or survival in an artificial stream channel. Roni (2003) examined response of reticulate sculpin (*Cottus perplexus*), torrent sculpin (*C. rhotheus*), larval lamprey (*Lampetra* spp.), and Pacific giant salamanders (*Dicamptodon ensatus*) to LWD placement in the coastal Pacific Northwest of the US and found higher densities of lamprey in treated stream reaches, but little difference for other nonsalmonid species.

Other studies on nonsalmonid fishes have examined changes in species diversity. Shields *et al.* (1993, 1995b) found large increases in fish species diversity following boulder weir placement in two streams in Mississippi (USA). Angermeier and Karr (1984) examined responses of 10 warm-water fishes to wood removal and placement in a small stream in Illinois (USA) and found more and larger fish in stream sections with woody debris. Jungwirth *et al.* (1995) examined instream structures on fishes in seven Austrian rivers and found increases in both fish biomass and species diversity when treated sections were compared with natural sections. Pretty *et al.* (2003) found no improvement in fish species abundance or diversity in 13 English streams treated with flow deflectors or artificial riffles. However, bullhead (*Cottus gobio*) and stone loach (*Barbatula barbatula*) tended to be more abundant in rehabilitated riffles than control areas.

Monitoring the response of the entire fish community is important in determining both project effectiveness and whether instream techniques are restoring fish

TABLE 11
Studies examining response of non-salmonids or entire fish community to placement of instream structure.

Study	Techniques Examined	Diversity/Richness	Other
Lyons and Courtney, 1990 (reviewed 20 studies)	Various boulder, and bank rotection structures; channel meandering	Three studies reported increase in fish	Nine studies reported increase in abundance (17 studies examined fish)
Roni, 2003	Various wood placement techniques	No difference	Increase in juvenile lamprey, no response for sculpin or salamanders
Shields et *al.*, 1993	Boulder weir placement to aggrade incised channel	Doubled	Approximate tenfold increase in fish biomass after treatment
Shields *et al.*, 1995a,b	Boulder weir placement to aggrade incised channel	Increased	Threefold increase in abundance and median fish size increased
Pretty et *al.*, 2003	Deflectors and stone riffles	No difference between rehabilitated and control	No difference in total abundance though bullhead and stone loach increased slightly
Angermeier and Karr, 1984	Placement of woody debris	No difference	Increased abundance, and size
Lonzarich and Quinn, 1995	Cover placement in artificial channel	Higher in pools with structure	Higher survival of water column species in pools with cover, but no difference in growth among species
Jungwirth et *al*.,1995	Removal of bank protection and installation of groins and bedfalls	Increased	Increased abundance and biomass
Linlokken, 1997	Rock weirs	No change	Percentage of minnows and brown trout increased, bullheads decreased

communities rather than just manipulating habitat for individual species. The studies discussed above suggest three major findings. First, in very diverse fish communities the increase and habitat complexity provided by instream habitat structures can lead to an increase in diversity. Second, projects designed for salmonids may have little effect on other species. Finally, little work has been done on nonsalmonid fishes and additional monitoring and evaluations considering all members of the fish community are needed.

Response to different techniques – Most instream rehabilitation projects involve the use of a variety of techniques to improve habitat, making comparison of different techniques difficult, and few studies have attempted to compare effectiveness among techniques. While we review them here, the results should be viewed with caution, as the effectiveness of a particular technique at a particular project can result from a variety of factors including project planning, construction, and recovery or integrity of watershed conditions and processes. Binns (1999) reported larger increases of trout numbers in constructed plunge pools or log weirs (129 percent increase) than at logjams (69 percent increase) and rock weirs (66 percent gain), but cautioned that many techniques were employed at various sites. Avery (2004) attempted to compare effectiveness among structure types using several fish success metrics and suggested that cover, current deflectors, and beaver dam removal achieved the best success ratings for resident trout. Slaney *et al.,* (1994) and Van Zyll De Jong *et al.,* (1997) provided more detailed comparisons of techniques within a given stream. Slaney *et al.,* (1994) examined rootwad, floating cribs, pseudo beaver lodges and debris catchers in the Keogh River, British Columbia, and found higher levels of juvenile Chinook and steelhead in treated than untreated areas, with debris catchers appearing to be the most cost-effective. Van Zyll De Jong *et al.,* (1997) examined the effects of three types of structures in a Newfoundland, Canada, stream and found increased Atlantic salmon (age 0+, age 1+, and age 3+) at boulder cluster sites, increased density of both brook trout and juvenile Atlantic salmon with v-weir placement, and increased age 0+ Atlantic salmon at half log (cover log) sites.

Other studies have found a strong correlation between the numbers of structures placed or amount of physical habitat change and fish response (i.e. Kennedy and Johnston, 1986; Roni and Quinn, 2001a). Kennedy and Johnston (1986) examined placement of stones in shallow areas (riffles) to improve salmon habitat in drainage maintenance (channelizing and periodic dredging of stream) and found a strong positive correlation between the number of stones placed in riffles and the juvenile Atlantic salmon density, while Roni and Quinn (2001a) found a positive correlation between amount of woody debris placed in the channel and juvenile coho salmon response to instream improvement. Roni and Quinn (2001a) found that the type of structure was not as important as whether a significant change in physical habitat occurred. This may also be the reason why projects in highly simplified streams seem to show the largest biological response. Moreover, as previously indicated, the success or failure of particular techniques at creating physical change often has to do with the appropriateness of the technique for that geomorphic situation and whether other larger factors such as water quality and riparian condition have been addressed.

Fish movement and watershed-scale evaluations—Examination of changes in fish abundance at instream enhancement projects can be complicated by immigration and emigration of fishes from nearby habitats or watersheds or by the effects of instream structures on fish movement. This is particularly evident for highly migratory fishes such as salmonids. For example, Riley and Fausch (1995) demonstrated an increase in abundance and biomass of three species of trout associated with structure installation in Colorado (USA) streams; however, a subsequent analysis of the same projects demonstrated that increases in biomass and abundance were strongly influenced by movement of trout into treated stream reaches (Gowan and Fausch, 1996). In contrast, Roni and Quinn (2001b) found little movement of juvenile salmonids between treatment and nearby reference reaches in a small Western Washington stream. Rinne (1982) reported reduced movement of gila trout *O. gilae* following the placement of high (>0.5 m) log dam structures. However, these studies did not examine population level responses but only local shifts in abundance.

The question of whether instream enhancement increases fish production or simply shifts fish distribution should be an important component of project evaluation (Gowan *et al.*, 1994; Frissell and Ralph, 1998; Roni and Quinn, 2001b). Underlying this is the need to examine both physical and biological responses at a population or watershed scale. Few studies have examined whole watershed response to habitat modifications. Solazzi *et al.*, (2000) found significantly higher abundance of juvenile coho following treatment. Conversely, Reeves *et al.*, (1997) examined fish response to a suite of rehabilitation efforts in the Fish Creek Watershed in Oregon, and found little evidence for increased fish production at a watershed scale. The difference in these two watershed-scale studies is likely a result of both differences in study design and changes in physical habitat due to rehabilitation actions.

While most instream projects occur at a site or reach scale, these projects may produce responses or affect responses of physical habitat and fish production in downstream reaches, in adjacent habitats, or throughout a watershed. This has been a particular concern for highly migratory fishes such as salmonids (Northcote, 1992), which typically have seasonal habitat preferences (Nickelson *et al.*, 1992; Roni, 2002). Thus, changes in one stream reach may affect salmonid abundance in adjacent stream reaches (Gowan and Fausch, 1996; Kahler *et al.*, 2001; Roni and Quinn, 2001b). Assessing the biotic responses and the physical responses at a watershed scale is arguably more important (and more difficult) than examining reach-scale responses such as changes in local fish abundance. Both the costs and the difficulty in implementation have made examining the effect of individual or multiple projects on physical habitat, fish populations, and other biota at a watershed-scale relatively rare. Yet watershed-scale monitoring and evaluation is a particularly important perspective to consider and is needed to thoroughly adequately understand fish responses to habitat rehabilitation.

Macroinvertebrate response—Equally important to monitoring responses of fishes to habitat enhancement is the need to examine the responses of aquatic macroinvertebrates, which are an important food source for fishes and highly sensitive to habitat alteration and disturbance (Merritt and Cummins, 1996; Karr and Chu, 1999). Several authors have considered the response of macroinvertebrates as a component of their monitoring strategy, if not focusing exclusively on them (Table 12). Early investigations by Tarzwell (1938) and Gard (1961) demonstrated an increase in macroinvertebrates following instream enhancement. More recently, Gortz (1998) detected an increase in some species of macroinvertebrates following placement of instream structures and Wallace *et al.*, (1995) found changes in functional feeding groups only within habitats altered by wood placement. Harper *et al.*, (1998) and Ebrahimnezhad and Harper (1997) examined artificially constructed riffles in a lowland English streams and found that constructed shallow riffles (< 30 cm) had more diverse macroinvertebrate communities than deeper constructed riffles and communities more similar to natural control riffles. Conversely, Hilderbrand *et al.* (1997), Laasonen *et al.* (1998), Larson *et al.* (2001), Brooks *et al.* (2002), Black and Crowl (1995) and Roni (unpublished data) detected no difference of macroinvertebrate density or diversity in enhanced and untreated stream reaches. These studies suggest that macroinvertebrates were likely limited by primary productivity rather than habitat complexity. Several studies have examined the recovery time of macroinvertebrates following instream rehabilitation efforts (Table 12). Tikkanen *et al.* (1994) found only short-term impacts of habitat rehabilitation efforts (boulder placement) on macroinvertebrate abundance. These disparate results may be the result of differences in scale of measurement, metrics examined, differences

TABLE 12
Summary of studies examining macroinvertebrate response to instream habitat enhancement. All studies examined response at a reach or individual habitat unit scale.

Reference	Structure types	Diversity/Richness	Abundance/Biomass
Tarzwell, 1937	Log weirs and deflectors	Not reported	Increased biomass, 3 to 9 higher after rehabilitation
Tarzwell, 1938	Log weirs and other structures	Not reported	Higher biomass in constructed versus natural pools
Gard, 1961	Log weirs and dams	Not reported	5 to 8 times higher biomass following structure placement and pool creation
Gerhard and Reich, 2000	Woody debris addition	Increased	increased
Black and Crowl, 1995	Woody debris addition	No significant change	NA
Wallace *et al.*,1995	Woody debris addition	Not reported	Abundance and biomass of scrapers and filters decreased, collectors increased, no change in overall shredder biomass
Hilderbrand *et al.*,1997	Woody debris additions	Not reported	No change in overall abundance of Ephemeroptera while other families decreased
Ebrahimnezhad and Harper, 1997, Harper *et al.*,1998	Constructed riffles	Similar in natural and constructed shallow riffles, lower in deeper artificial riffles and similar to deep riffles in channelized reaches	Not reported
Gortz, 1998	Boulders, gravel and stream concentrators	Similar in treatment and reference (natural) reaches	Clean water species more abundant in restored reaches
Laasonen *et al.*,1998	Boulder dams and flow deflectors	No significant difference between treatment and control reaches	No significant difference between treatment and control reaches
Muotka and Laasonen, 2002	Boulder dams and flow deflectors	Not reported	Algae consuming shredders increased
Muotka *et al.*, 2002	Boulder dams and flow deflectors	Small but insignificant differences between treatment and control reaches	No significant difference between treatment and control reaches
Larson *et al.*, 2001	Woody debris placement	No difference	No difference
Brooks *et al.*, 2002	Modification of substrate size in riffles (addition of boulders and cobble)	No difference	No difference
Tikkanen *et al.*,1994	Channel enlargements, structure placement	No changes detected	Relative quick recovery to levels in following rehabilitation disturbance
Roni (Unpublished, data from 23 streams)	Weirs and randomly placed logs	No significant difference between treatment and control reaches in diversity, Index of Biotic Integrity, or other metrics examined	No significant difference between treatment and control reaches
Negishi and Richardson, 2003	Boulder clusters	No changes	Increase

in project objectives and physical habitat change, or perhaps that macroinvertebrates are not a good indicator of success of instream habitat enhancement for fish.

A few studies have examined the recovery of macroinvertebrates following construction of instream enhancement efforts. Muotka *et al.* (2002) found that macroinvertebrate communities recovered within 4 to 8 years following disturbance with heavy equipment during construction of instream rehabilitation actions. Biggs *et al.* (1998) indicated that macroinvertebrate communities and aquatic macrophytes recovered within a year or more following intensive rehabilitation activities on the Brede, Cole, and Skerne rivers in Denmark and the United Kingdom. Limited evidence from these studies suggests that construction impacts are relatively short lived and that organic matter retention increases following increases in habitat complexity and instream structure.

Effects on organic matter and aquatic vegetation – The biological response to instream rehabilitation has focused on fishes and macroinvertebrates, though these activities can influence other species as well as other stream processes. Wood and other instream structure help increase retention of organic matter (Bilby and Likens, 1980) and placed wood has been shown to function in a similar manner (Trotter, 1990; Gerhard and Reich, 2000). In addition, habitat complexity and the retention of organic mater and nutrient spiralling are important. Muotka and Laasonen (2002) found increased organic matter retention following boulder and other structure placement in eight Finish headwater streams, but moss cover decreased dramatically following rehabilitation efforts. Wallace *et al.* (1995) and Negishi and Richardson (2003) found increased organic matter retention following wood placement and boulder clusters, respectively. These studies suggest that instream structures may have an influence on watershed processes such as nutrient and organic matter retention and utilization.

A few other studies have examined the effects of instream structures on aquatic macrophytes. For example, O'Grady (1995) found differences in aquatic macrophyte diversity between reaches of two Irish streams following placement of boulders. These studies suggest some improvement in macrophyte composition and growth from instream rehabilitation likely from changes in velocity and depth, while other studies (e.g. Iversen *et al.*, 1993; Muotka and Laasonen, 2002) suggest disturbance due to construction of rehabilitation projects can lead to short-term impacts to aquatic macrophytes.

2.5.3 Conclusions – instream habitat structures

Our review suggests instream enhancement projects produce improvements in physical habitat and biota when implemented correctly and provides support for their continued use. However, given the variability in results for various species and structure types, the limited number of statistically rigorous studies, differential responses by different species or life stages, and the cost of instream enhancement projects, it is apparent that these projects should be undertaken with careful consideration of scale, watershed conditions and processes, and coupled with a rigorous monitoring programme. Several books are dedicated to the appropriate application and design of instream habitat rehabilitation and should be consulted (e.g. Slaney and Zoldakis, 1997; Cowx and Welcomme, 1998).

The following are key findings and factors to consider when implementing instream habitat projects.

- Placement of logs and log and boulder structures leads to improvements in physical habitat and local fish abundance when implemented correctly.
- Success of efforts is often tied to larger watershed-scale factors such as water quality, hydrology, sediment delivery, and riparian and upslope conditions.
- Gradient and stream power will influence the effectiveness of structures and many structural techniques are not appropriate in higher energy channels.
- The potential benefits of most instream structures may be short lived (< 10 years) unless coupled with riparian planting or other process based restoration activities that will lead to long term recovery of deficient processes.
- While placement of instream structures appears to be successful at increasing local fish abundance, particularly for salmonid fishes, results are highly variable by species and life stage as well as structure type.

- The most successful projects are those that create large changes in physical habitat and mimic natural processes.
- Watershed and population level responses to placement of instream structures for both physical habitat and biota are still largely unknown.

2.6 LAKE HABITAT ENHANCEMENT

2.6.1 Common lake habitat enhancement techniques

As with the placement of instream structures, the placement of artificial structures in lakes has been conducted since at least the early 1930s (Brown, 1986; Bolding *et al.*, 2004). Habitat enhancement in reservoirs is particularly common because of the lack of substrate complexity and cover (Vogele and Rainwater, 1975; Prince and Maughan, 1978; Wills *et al.*, 2004). The purposes of artificial structures in ponds, lakes, and reservoirs are similar to those for artificial reefs in the marine environment: to concentrate fish, increase harvest levels, and increase localized or overall biological production in the water body (Rountree, 1989; Bassett, 1994; Bohnsack *et al.*, 1997; Lindberg, 1997; Bolding *et al.*, 2004). In addition, structures are often placed in aquaculture ponds to increase fish production or in natural floodplain lakes and other habitats to attract fish to improve harvest (Welcomme, 2002). As no studies have examined physical effects, we summarize the biological effectiveness of structures in lakes.

Lake habitat enhancement structures can be categorized based on the types of material used and whether they are designed for cover or for spawning habitat. These include cover structures made of trees or artificial materials, addition of gravels or boulders, and planting of vegetation (Table 13; Figure 10). Below we briefly discuss each type and what is known about their effectiveness.

2.6.2 Effectiveness of lake habitat enhancement

The goals and effectiveness of structures placed in lakes and ponds to improve fish habitat have been periodically reviewed and summarized (Prince and Maughan, 1978; Brown, 1986; Tugend *et al.*, 2002; Wills *et al.*, 2004). For example, in a recent survey in the United States, Tugend *et al.* (2002) found that 75 percent of the lake habitat enhancement included placement of structures, while 25 percent were spawning structures or substrate manipulations.

Habitat enhancement structures can produce positive results through increased spawning recruitment and survival, refuge or cover from predation, shade and sites for orientation and schooling, can increase prey as well as feeding efficiency of adults, and provide increased harvest and fishing opportunities (Bolding *et al.*, 2004; Wills *et al.*, 2004). Prince and Maughan (1978) suggested that habitat structures in ponds and lakes increase both fish prey and production. However, a number of factors affect the

TABLE 13

Common pond, lake, and reservoir physical habitat enhancement techniques and their goals and effectiveness. See section on lake habitat enhancement for references.

Category	Common types of structures	Goal	Effectiveness
Cover structures	Logs, brush bundles, artificial reefs (tires, concrete, etc.), floating structures	Concentrate fish, provide cover for juvenile fish or predators, provide fishing opportunity	Positive response for many warmwater fishes and increased catch, many factors influence success for different species
Spawning habitat	Rock shoals, reefs, same structures as above	Create spawning habitat or cover near spawning habitat	Increased fish use detected in some studies, unclear if lead to increased production, much debate about effectiveness for walleye and lake trout
Shoreline vegetation	Planting during low water	Provide habitat during high water events, provide source of organic matter, and woody debris for in water cover	Known to increase cover, stabilize banks, thought to benefit fish but rigorous biological monitoring lacking

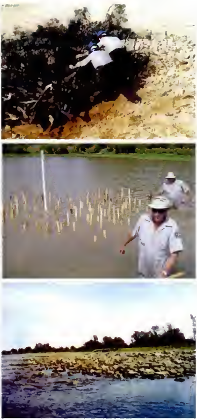


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FIGURE 10. Photographs of common lake habitat enhancement techniques used in North America including brush bundles (top), stake bed (middle), and rock shoal for walleye spawning habitat (bottom).

success of structures including the structure type, construction material, interstitial spaces, depth and location within the lake, lake morphometric characteristics, substrate complexity, presence of aquatic macrophytes, and the fish species and life stage examined (Walters *et al.*, 1991; Lynch and Johnson, 1989; Moring *et al.*, 1989; Rogers and Bergersen, 1999; Tugend *et al.*, 2002; Wills *et al.*, 2004; Bolding *et al.*, 2004). Moreover, fish use of structures varies daily (deil variation), seasonally, and with water temperature (Prince and Maughan, 1978; Moring and Nicholson, 1994). Similar to studies of artificial structures in the marine environment, the effect of these structures on fish population size or fish production has not been examined (see Hoff, 1991; Tugend *et al.*, 2002).

Brush bundles and log cover structures—Brush bundles, logs, and half-log structures are made of natural material and are typically placed on or suspended near the bottom (Brown, 1986). Brown (1986) and Tugend *et al.*, (2002) reviewed published evaluations of brush bundles and indicated that several studies have reported increased abundance and catches of fishes at treatment (brush structures) versus control sites (no brush added) particularly for centrarchids and cyprinids (Brown, 1986; Graham, 1992; Moring and Nicholson, 1994; Tugend *et al.*, 2002), but less so for other species such as yellow perch (*Perca flavescens*) (Moring *et al.*, 1989).

Cover logs or placement of whole logs or trees into lacustrine environments typically have the same goals as brush bundles. Similarly they have been shown to have a positive response for centrarchids and other warmwater fishes (Moring *et al.*, 1989; Wills *et al.*, 2004). For example, smallmouth bass (*Micropterus dolomieu*) numbers, nest density, and nest success improved following placement of half logs in both lakes and reservoirs (Hoff, 1991; Wills *et al.*, 2004). Submerged logs left over from logging operations supported five of 10 species found in reservoir in Maine, US (Moring *et al.*, 1989). There is also some evidence that a combination of brush, vegetation, and gravel placed in the littoral zone can benefit young of year pikeperch (*Stizostedion lucioperca*) even in the presence of a predator (Northern pike, *Esox lucius*) (Lapínska *et al.*, 2001).

While a number of studies have examined the response in North America lakes, placement of brush and logs into lakes and floodplain habitats to enhance fish production is also utilized in the developing world (Welcomme, 2002). For example, dropping of logs to rehabilitate designated fish sanctuaries has occurred in the Tonle Sap Great Lake in Cambodia (Thuok, 1998). Similarly, Vinci *et al.* (2003) described three Indian variations of placing bamboo and brush into floodplain ponds in India. Brush bundles or brush parks are widely used in floodplain lakes and ponds to enhance both native and cultured fish production in developing countries (Hoggarth *et al.*, 1999; Quasim, 2002). These are primarily installed to provide cover for fishes and many different modifications exist particularly for fish culture practices (Quasim, 2002; Figure 11). However, in some areas this practice is illegal or they are designed as refuges or fish sanctuaries and fishing in these areas is illegal. Studies of their effectiveness are rare, but evaluations of fish sanctuaries and brush parks in Bangladesh have shown them to be highly effective at both attracting fish and recovering endangered species (Poulsen and Hasan, 2004).

Cover structures made of artificial materials – A variety of artificial materials have also been used for creating cover and habitat complexity in lakes. Stake beds, concrete slabs or structures, plastic crates, boxes, and pipes and tire were the most commonly reported in the literature (Brown, 1986). Stake beds are typically composed of hundreds of stakes driven into the substrate or prefabricated onshore prior to installation. These artificial structures are designed to provide cover, holding, and in some cases spawning habitat.

The results for structures made from artificial materials have been less consistent than for brush bundles or logs. A handful of studies have reported positive results from placement of highly artificial structures in lakes. Moring and Nicholson (1994) examined fish use of cinder blocks, tire bundles, and brush bundles and found higher levels of pumpkinseed sunfish (*Lepomis gibbosus*), chain pickerel (*Esox niger*), brown bullhead (*Ameiurus nebulosus*), common shiner (*Luxilus cornutus*), and golden shiner (*Notemigonus crysoleucus*) and found higher densities of all species near artificial structures than sites without. Sampling before and after placement of cinderblock and log structures, Graham (1992) found higher numbers of small bluegill (< 15cm) in Lake Anna, Virginia (USA) following treatment. Prince and Maughan (1979a,b) found that tire reefs significantly concentrated both bass and sunfish in another Virginia lake. Wege and Anderson (1979) reported higher growth rates of largemouth bass (*Micropterus salmoides*) and higher catch rates of both bluegill and largemouth bass in ponds treated with tires and stakes beds or brush bundles than control ponds.

In contrast, other studies have found no change in fish use or higher fish use of structures composed of natural materials versus artificial materials. Wills *et al.* (2004) examine plastic box shaped shelter devices as a habitat enhancement device and found no change in fish use before and after placement of plastic cover structures (Aqua Cribs). Pardue and Nielsen (1979) examined tilapia (*Tilapia* spp.) and bluegill (*Lepomis macrochirus*) response to pine board structures in small experimental ponds and found little evidence that structures increased fish biomass. Wilbur (1978) examined clay pipes and cement blocks with brush as fish attraction devices and found several warmwater species used the structures. However, structures that contain trees or brush were preferred by fishes over structures made of purely artificial materials (Wilbur 1978).

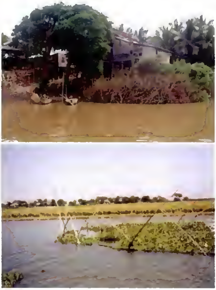


FIGURE 11. Brush parks or "samrah". A typical samrah brush park on the Tonle Sap River, Cambodia with bushes and branches gathered together with submerged poles or fences to prevent them being carried away (top). Highly effective at attracting fish, they are illegal in many areas. Modification on the samrah in Bangladesh using water hyacinth (bottom). Here the brush park is legally used as a fish sanctuary where fishing is not allowed.

Rogers and Bergersen (1999) examined a suite of highly artificial structures fastened to steel frames and found higher largemouth bass use of structures containing trees and no use of any of the artificial structures by northern pike. Johnson and Lynch (1992) found fish use (bluegill sunfish and white crappie, *Pomoxis annularis*) and angler success higher at brush bundle and evergreen sites than stake bed structures. Moreover, Hunt *et al.* 2002) found that artificial structures were only successful at providing cover and nesting habitat for largemouth bass if they were similar to natural habitat (logs or brush) and placed in the littoral zone where bass typically spawn and woody debris was historically most abundant. The advantages of log structures and brush bundles over structures made of artificial materials apart from their apparent higher level of success is that they mimic what may have been found in many lakes prior to lakeshore development and removal of riparian vegetation.

While only a few studies have examined cost-effectiveness, it appears that more natural materials, such as brush, trees, logs, or rock and gravel, are not only more likely to be utilized by fishes but are more cost-effective. This is in part a result of the high cost of fabricating many structures composed of artificial material (Johnson and Lynch, 1992). Studies such as Paxton and Stevenson (1979) and Wege and Anderson (1979) have reported higher catch rates at artificial structures than natural unstructured areas and suggested that artificial structures may lead to increased exploitation from fish harvest rather than an increase in population.

Similar to studies on fish aggregation devices in the marine environment (Lindberg, 1997), logs and other floating material have also been demonstrated to attract fishes in lakes (Helfman, 1979). For example, Helfman (1979) reported higher numbers of bluegill, black crappie, and golden shiner were more abundant near or under floating structures, while several other cyprinids and centrarchids showed little affinity for structures. Floating structures provide cover for both schooling fishes and predators; thus it is unclear which species or trophic levels benefits most from these types of structures. As discussed in the section on instream rehabilitation techniques, rigorous studies are needed to determine whether placement of structures in lentic habitats leads to increased production or simply just concentrates fish.

Artificial shoals or spawning areas—Creating spawning areas within lakes typically involves the placement of gravel, cobble, boulders, or concrete rubble to create suitable areas. As previously discussed, many cover structures such as brush bundles and logs are designed to provide cover for both juveniles and spawning fishes (Tugend *et al.*, 2002; Wills *et al.*, 2004). In the Great Lakes of North America, artificially constructed reefs or spawning shoals account for much of the lake trout (*Salvelinus namaycush*) spawning areas in Lakes Ontario and Michigan (Fitzsimons, 1996). In a recent review of research on use of artificial reefs in the Great Lakes, Fitzsimons (1996) reported higher levels of eggs, fry, and young-of-year at artificial reefs and spawning areas. However, success of naturally produced lake trout in the Great Lakes is poor and other factors besides spawning appear to be limiting the success of artificial reefs at increasing fish production.

Artificial reefs can provide spawning habitat for a number of warm-water fishes such as sunfish (*Lepomis macrochirus*) and walleye (*Stizostedion vitreum vitreum*) and provide for increased catch rates for yellow perch and other species (Brown, 1986; Kelch *et al.*, 1999). Geiling *et al.* (1996) reviewed the effectiveness of creation of artificial reefs to create walleye spawning habitat in the Great Lakes and found that it rarely increased adult walleye abundance and suggested that this was not a successful technique for this species. Similarly in a review of 20 rock placement projects designed to improve walleye habitat in Wisconsin lakes (USA), Neuswanger and Bozek (2004) reported that 85 percent showed no signs of increased fry production. However, the success or failure of artificial spawning areas in lakes or rivers at increasing fish populations is driven in part by their location relative to rearing areas for juveniles (Jones *et al.*, 2003). The information on the effectiveness of spawning habitat enhancement in lakes is limited but appears to only be successful if they are carefully designed and placed with consideration of the biology and ecology of the species of interest.

Shoreline vegetation—The planting of lakeshore vegetation (riparian vegetation) provides a number of benefits to lakes including shade, source of organic matter, protection from wind (which can change mixing), bank protection, and food (terrestrial insects), and is a source for brush, logs and other cover structures. In addition, lakes or reservoirs with seasonally inundated areas are planted with vegetation which provides good habitat for young of year fish such as bass (Brouha and von Geldern, 1979; Brown, 1986). Little biological evaluation has occurred following planting of lake shores, but the planting of water tolerant vegetation such as *Salix* spp. helps stabilize banks and prevent erosion, which may also improve fish habitat near structures and in the waterbody.

Concentration versus increased harvest—In both the freshwater and marine environment, there is a concern that placement of structures is only leading to concentration of fish and increased harvest rates rather than increased production (Roni and Quinn, 2001b). Paxton and Stevenson (1979), Wege and Anderson (1979) and others have reported higher catch rates at artificial structures than natural unstructured areas and suggested that artificial structures may lead to increased exploitation from fish harvest rather than an increase in population. Additional work is needed to clarify whether this is true or not.

2.6.3 Conclusions — lake habitat enhancement

The existing literature on addition of structures to lakes and ponds to improve habitat suggests that natural structures such as logs and brush bundles lead to increases in local abundance of some species. The relative success of structures made of natural materials is also supported by ecological studies which demonstrate the importance of riparian areas and woody debris along the shoreline and littoral zone of lakes (see Christensen *et al.*, 1996). The literature also indicates that placement of habitat structures in lakes is an effective strategy for providing refuge areas, fishing opportunities and has a long history of use in aquaculture. We make the following recommendations based on our review of the literature.

- Addition of logs, wood, and brush bundles is a suitable form of habitat enhancement in areas were cover is limiting fish production.
- Structures made of artificial materials are not as effective as those constructed of logs or brush.
- Spawning reefs constructed of gravel, rock and boulders have not been demonstrated to be and effective method for increasing production.
- A number of factors effect project success including: fish species, limiting factors, location of structures (depth, vicinity to shoreline, proximity to other important habitats), predators-prey interactions, increased harvest due to fish concentration, and whether the structures mimic natural habitats.
- Additional studies are needed to determine population level responses to habitat enhancement in lakes.

2.7 NUTRIENT ENRICHMENT

High levels of nutrients (N and P) from agricultural runoff (both fertilizers and animal waste), storm water inputs, and treated and untreated sewage have lead to high levels of nutrients and water quality problems in many lakes, streams, and other freshwater habitats (Kiffney *et al.*, 2005). Conversely, in many oligotrophic waters in temperate and arctic latitudes, the addition of organic and inorganic nutrients (N and P) has been used as a method to increase primary and secondary productivity and presumably fish growth and abundance (Benndorf, 1990; Kiffney *et al.*, 2005). Nutrient additions are also widely used in aquaculture in floodplain lakes and ponds to boost fish production.

2.7.1 Nutrient enrichment techniques

A handful of techniques have been developed to enrich nutrient levels and boost productivity of waters including the addition of organic (fish carcasses) and inorganic nutrients (typically N and P). For aquaculture purposes, this often includes diversion of nutrient rich agricultural run off, addition of inorganic nutrients, and addition of manure or other solid organic material (Welcomme and Bartley, 1998). The recent popularity of nutrient enrichment in North America has lead to some new developments for applying these nutrients including "fish blocks", which are produced using fish feed or bone and flesh waste from fish processing plants to time release fertilizers specially designed for stream applications (Sterling *et al.*, 2000; Wipfli *et al.*, 2004; Kiffney *et al.*, 2005). While nutrient enrichment techniques are sometimes termed restoration, they are in essence a form of enhancement that is more often than not used in an attempt to increase the productivity of a water body above historical levels. However, sustaining this level of production is typically dependent on continued addition of nutrients (Guthrie and Peterman, 1988; Gross *et al.*, 1998;).

The addition of nutrients to streams and a lesser extent lakes has become more common place in western North America following a number of studies that demonstrated the importance of anadromous fishes to the productivity of freshwater ecosystems (Krokhin, 1975; Cederholm *et al.*, 1999; Roni *et al.*, 2002; Kiffney *et al.*, 2005). For example, it is estimated that reductions in returning salmon to the Pacific Northwest coastal streams has reduced marine derived N and P delivered by returning adult salmon to six to seven percent of historic levels (Gresh *et al.*, 2000). Studies for other anadromous fishes such as smelt (*Osmeridae*), shad (*Alosa* spp.), Atlantic salmon, and sea trout have demonstrated similar results in tributaries to the North Atlantic (Durbin *et al.*, 1979; Garman and Macko, 1998; Lyle and Elliott, 1998).

2.7.2 Effectiveness of nutrient enrichment

Lakes –The effectiveness of these techniques has been examined in numerous studies in both lakes and streams in North America and Europe, though much of the work has occurred in lakes. The experimental manipulation of N and P levels to increase primary and secondary production in lakes has been occurring for many years in Alaska (Kyle, 1994; Mazumder and Edmundson, 2002), midwestern US (Carpenter *et al.*, 1996; Cottingham and Knight, 1995; Vadeboncoeur *et al.*, 2001; Essington and Houser, 2003), western United States (Sarnelle, 1992; Cottingham and Carpenter, 1998; Wurtsbaugh *et al.*, 2001; Budy *et al.*, 1998), British Columbia, Canada, (Hyatt and Stockner, 1985; Stockner and MacIsaac, 1996; Ashley *et al.*, 1997; Johnston *et al.*, 1999; Bradford *et al.*, 2000), Northwest Territories (Welch *et al.*, 1988; Jorgenson *et al.*, 1992), eastern Canada (Clarke *et al.*, 1997; Mazumder *et al.*, 1988), Denmark (Jeppesen *et al.*, 1991), Sweden (i.e. Bjoerk-Ramberg, 1983; Jansson, 1984; Holmgren, 1984; Bjoerk-Ramberg and Anell, 1985; Jansson *et al.*, 2001), Norway (Langeland, 1990), and Japan (Ishida and Mitamura, 1986). Increases in nutrients, phytoplankton (chlorophyll a) and primary production, zooplankton, and benthic macroinvertebrates have been demonstrated in many of these studies.

Fish response to lake enrichment has been more variable and difficult to measure. The vast majority of studies on fish response to lake enrichment have been for sockeye salmon (*Oncorhynchus nerka*) and most have demonstrated increased growth, juvenile survival, and sometimes adult sockeye returns (Hyatt and Stockner, 1985; Ashley *et* *al.*, 1997; Kyle, 1994; Stockner and MacIsaac, 1996; Budy *et al.*, 1998; Bradford *et al.*, 2000; Mazumder and Edmundson 2002). A few studies have reported positive results for other species. Johnston *et al.* (1999) examined the effects of inorganic fertilizers on rainbow trout in a British Columbia lake and found increased growth, reproductive output, but not survival. Essington and Houser (2003) examined yellow perch (*Perca flavescens*) response to nutrient enrichment and found increased growth and reduced mercury concentrations. Other studies that have focused on examining the effects fish (predator) additions or removals have on food webs and ecosystem components (e.g. Mazumder *et al.*, 1988; Benndorf, 1990; Carpenter *et al.*, 1996; Cottingham and Carpenter, 1998). Studies of nutrient additions of aquaculture ponds and reservoirs in China and Russia have also demonstrated large increases in plankton, benthos, and fish yield following addition of manure or other organic nutrients (Lu, 1992; Welcomme and Bartley, 1998).

Based on studies of fish as well as primary and secondary productivity, the response of a given lake to fertilization depends on a variety of factors including but not limited to: zooplankton and fish species present (Jeppesen *et al.*, 1991; Benndorf, 1990; Carpenter *et al.*, 1996), light levels (Bjoerk-Ramberg, 1983; Benndorf, 1990), stratification (temperature) and depth (Bjoerk-Ramberg, 1983; Jansson, 1984; Wurtsbaugh *et al.*, 2001), whether benthos were sampled (Bjoerk-Ramberg and Anell, 1985; Vadeboncoeur *et al.*, 2001), whether the system was P or N limited (Holmgren, 1984; Jansson *et al.*, 2001), number of years of fertilization (Jorgenson *et al.*, 1992), or top down versus bottom up control of biotic communities (Langeland, 1990; Stockner and Shortreed, 1994).

Streams—The addition of both organic (fish tissue) and inorganic nutrients on streams and their biota does not have as long a history as that for lakes, but there still are several studies that have examined the effects of these two techniques on stream biota (Table 14). Johnston *et al.* (1990), Ward (1996), and McCubbing and Ward (1997, 2000), in one of the largest and longest term studies on addition of inorganic nutrients to a stream, have detected increases in periphyton, macroinvertebrate abundance, and increases in juvenile coho salmon and steelhead growth and condition. Ashley and Slaney (1997) summarized the results of case studies of inorganic nutrient additions on five different British Columbia watersheds and found increased periphyton, invertebrate biomass, and fish growth following nutrient additions. Wipfli *et al.*, (2003, 2004) in a series of studies in both natural and artificial streams in Southeast Alaska, found increased condition, growth, and production following placement of both salmon carcasses and carcass analogs (artificial carcasses made from fish tissue). Following addition of inorganic phosphorous to an Arctic tundra river, Deegan and Peterson (1992) found an increase in juvenile and adult arctic grayling growth. These studies all suggest positive initial responses to addition of inorganic nutrients to oligotrophic streams.

Addition of organic nutrients in the form of fish carcasses has also been examined in anadromous fish streams in Alaska, British Columbia, Washington and Minnesota. For example, Schuldt and Hershey (1995) found increased P and periphyton following addition of salmon carcasses to a Minnesota stream. In a laboratory experiment, Minakawa *et al.* (2002) found higher growth of caddis flies on salmon carcasses than on natural leaves. Wipfli *et al.* (1998; 1999) and Chaloner and Wipfli (2002) found increased chlorophyll a, macroinvertebrates, and growth rate of salmonids following additions of carcasses in artificial and natural stream channels. Studies using stable isotope analysis have demonstrated the importance of salmon carcasses in aquatic food webs for primary productivity, macroinvertebrates, fishes and even riparian vegetation and tree growth (Bilby *et al.*, 1996; Helfield and Naiman, 2001). While additional study is needed, these studies all suggest that the placement of salmon carcasses or an increasing in spawning fish can increase fish growth and possibly survival.

TABLE 14

Summary of published studies on nutrient enrichment in streams

Stream	Nutrient type	Primary and secondary production	Fish	References
Keogh River, B C Canada	Inorganic N & P	Increased periphyton standing crop	Increased juvenile salmonid growth and survival	Johnston *et al*, 1990, Ward, 1996, McCubbing and Ward, 1997, 2000
Salmon, Big, A400 and Wasberg Creeks, Washington State, USA	Salmon carcasses	NA	Increase in juvenile salmonid density, increase in marine derived nutrients in fish tissue	Bilby *et al*, 1998
Salmon, Adam, Big Silver, Mesilinka Rivers River	Inorganic nutrients	Increase in chlorophyll a and macroinvertebrate biomass	Increase in juvenile salmonid density and biomass	Ashley and Slaney, 1997
Fish Creek, Alaska	Salmon carcasses	Increased growth rate of macroinvertebrate collectors, but not consistently for other groups	NA	Chaloner and Wipfli, 2002
Kuparuk River, Alaska	P (phosphoric acid)	Chlorophyll increased (five-fold) Caddis flies abundance was higher in treatment reaches, no differences for other families	Increased growth rates of adult and young-of-year arctic grayling, increased neutral, lipid storage	Deegan and Peterson, 1992, Deegan *et al*, 1997
Several artificial and natural stream channels in Southeast Alaska	Addition of salmon carcasses, and salmon analog (processed fish block)	No difference in artificial channels, but significantly higher in natural stream, increased macroinvertebrate abundance	Increased growth, condition factor, and production of salmonid fishes	Wipfli *et al*, 1998, 1999, 2003, 2004
Artificial channels	Slow release inorganic phosphate fertilizer	Increased periphyton & primary productivity	NA	Sterling *et al*, 2000
Griffin Creek	Salmon carcasses	Increased macroinvertebrate growth (Trichoptera)	NA	Minakawa *et al*, 2002
Stewart and French rivers (Minnesota)	Salmon carcasses	Increased phosphorus, nitrogen, periphyton biomass	NA	Schuldt and Hershey, 1995

2.7.3 Conclusions – nutrient enrichment

The addition of organic and inorganic nutrients into lakes and ponds has a relatively long history with variable success depending on a number of ecological factors. Nutrient enrichment in streams has a much shorter history, but early studies suggest very positive results. While additional studies are needed particularly for stream nutrient enrichment, we provide the following conclusions and recommendations.

- The addition of inorganic and organic nutrients to oligotrophic streams and lakes can lead to increased growth and production of algae and zooplankton and in some cases increased fish growth.
- Continued addition of nutrients or an increase in natural nutrient delivery (recovery of depressed anadromous fish runs) is needed to maintain elevated production.
- Many factors affect the success of projects and need to be evaluated prior to nutrient enrichment including: base line nutrient status (what nutrients are limiting productivity?), species composition of plankton, macroinvertebrates and fishes (top down versus bottom up control), physical limnology of lake (e.g. depth, stratification, temperature, light).

Additional discussion of determining whether nutrient enrichment is appropriate and methods for monitoring and evaluating responses are provided in Ashley and Slaney (1997) and Kiffney *et al.* (2005).

2.8 MISCELLANEOUS HABITAT REHABILITATION METHODS

There are several less common methods of habitat rehabilitation including: beaver reintroduction or removal, streambank debrushing, bank protection, and habitat protection. We discuss these here as they do not easily fit into the categories we have described and because may or may not be habitat rehabilitation, depending on the project goals.

2.8.1 Beaver reintroduction or removal

Beaver dams alter the hydrology and geomorphology of stream systems and affect habitat for fishes as well as fish diversity (Snodgrass and Meffe, 1998; Pollock *et al.*, 2003). They also influence the rates of groundwater recharge and stream discharge, retain enough sediment to cause measurable changes in valley floor morphology, and enhance stream habitat quality for many fishes (Pollock *et al.*, 2003). Historically, beaver dams were frequent in small streams throughout most of the Northern Hemisphere. The cumulative loss of millions of beaver dams has affected the hydrology and sediment dynamic of stream systems. The cumulative hydrologic and geomorphic affects of the loss of millions of wood structures (beaver dams) from small and medium-sized streams are not entirely known. This is particularly important in semiarid climates, where the elimination of beaver and beaver dams has likely exacerbated effects of other land use changes, such as livestock grazing, and possibly accelerated incision and the subsequent lowering of groundwater levels and ephemeralization of streams (Pollock *et al.*, 2003).

Beaver reintroduction has been proposed as a method of restoring the ecological functions described above and has in fact been attempted on a limited basis in the Europe, Russia, Mongolia, and North America. Beaver reintroductions in the USA suggest that rapid recolonization, dam construction, and changes in physical habitat occur following reintroduction, assuming the animals are not harvested or consumed by predators (Apple, 1985; Albert and Trimble, 2000; McKinstry *et al.*, 2001; McKinstry and Anderson, 2002). For example, McKinstry and Anderson (2002) reported successful reintroduction of beavers in 13 of 14 sites. They indicated that best success was gained with beavers greater than 2 years old, as mortality rates among younger beaver were extremely high. Merely reducing or banning of commercial or recreational harvest (trapping) of beaver has lead to them slowly recolonizing many areas in the United States. Studies in Poland have also shown that beavers rapidly colonize habitats following beaver reintroduction (Zurowski and Kaspercyzk, 1988).

While beavers, beaver dams, and associated ponds are known to provide a number of benefits to riparian areas, floodplains, and aquatic ecosystems and particular benefits for juvenile Pacific salmon (Pollock *et al.*, 2004), some fisheries management programmes have recommended removal of beavers and beaver dams to enhance resident trout populations. Unpublished studies in the mid-twentieth century suggested this as an effective technique for improving brook trout habitat in Wisconsin (USA) streams (Avery, 2004). In a review of 103 trout habitat improvement projects conducted in Wisconsin between 1953 and 2000, Avery (2004) reported that beaver dam removal resulted in the highest success rate and largest increase in brook and brown trout numbers of any project type they examined. Avery did not examine nonsalmonid fishes or other ecological factors and his results suggest that beaver dam removal may be effective for a narrow set of objectives such as increasing resident brook trout numbers. However, if the goals are ecosystem recovery or restoration, then reintroduction or protection of beavers appears to be more successful strategy than removal and would benefit many native fishes and other biota.

2.8.2 Bank protection (riprap and bioengineering)

Riprap (large rock placed to protect the bank), riparian plantings and other bank protection methods are often incorporated as part of habitat rehabilitation efforts though their objectives are not always to improve fish habitat. Riparian rehabilitation activities that utilize living materials such as trees and shrubs for bank protection and other forms of fluvial geomorphic control are termed bioengineering (Schiechtl and Stern, 1997). Often the purpose of such activities is to rapidly increase both aboveground and belowground biomass such that the erosion of the underlying substrate is minimized during floods. Bioengineering frequently involves the use of both living and nonliving materials to create a desired feature. Thus gabions, groins, sills and other bank protection structures are planted with living material such as live cutting of willow to restore certain riparian functions. The numerous bioengineering techniques that can be applied towards bank protection are described in detail elsewhere (Gray and Sotir, 1996; Schiechtl and Stern, 1996, 1997). It is important to note that bank protection is a mitigation strategy to protect infrastructure or areas of human uses. It is clearly effective strategies for these objectives and may prevent erosion, but improving habitat is typically a side benefit of these techniques.

The effectiveness of bank hardening, planting, and bioengineering techniques at stabilizing banks has been relatively well documented (RSPB *et al.*, 1994; Schmetterling *et al.*, 2001). In some highly degraded systems, bank protection may prevent further erosion and provides benefits for fishes (O'Grady *et al.*, 2002; Schmetterling *et al.*, 2001). However, it does not provide habitat for multiple life stages and can disrupt natural channel processes such as channel migration and development of natural riparian vegetation (Schmetterling *et al.*, 2001). Moreover, higher diversity of fishes in natural (woody debris, sand and grass banks) versus artificial habitats (riprap banks, bridge piling, groins) has been documented (Madejczyk *et al.*, 1998; Peters *et al.*, 1998; Schmetterling *et al.*, 2001). Those bank protection techniques that include bioengineering appear to be more beneficial to fisheries resources than hard structures. For example, Peters *et al.* (1998) found that bank protection with woody debris had higher fish densities than those that use only rock, but still had lower fish densities and woody debris levels than natural stream banks. These studies suggest that when necessary, bank protection that incorporates woody debris, willow plantings or similar items (bioengineering) appear be more beneficial to fisheries resources than hard structures, though they may not achieve the benefits of more natural stream banks. In our view bank protection is not really a form of habitat improvement and should be used cautiously in habitat rehabilitation activities.

2.8.3 Brush removal

Debrushing a stream or bank is another technique that seems to be in direct contrast to many other efforts focused on restoring riparian conditions and streambank rehabilitation. A major assumption is that removal of dense brush that shades the stream will lead to an increase in macroinvertebrates and thus fish production, which is in part supported by results of previous studies on logging (see Hicks *et al.*, 1991 for a review). Brush removal and trimming of vegetation are also common methods for manipulating habitats for birds, wildlife, and sometimes to control localized flooding (RSPB *et al.*, 1994). Brush removal is also sometimes used as a method for controlling invasive exotic riparian species and allow for recolonization of native vegetation (see section on riparian rehabilitation). The cutting of weeds to form a sinuous channel while leaving bank brush and vegetation is a technique for fisheries rehabilitation common in Denmark and other countries (Iversen *et al.*, 1993).

The effects of brush removal on fisheries resources have been inconsistent. O'Grady (1995) found higher level of Atlantic salmon age 1+ and brown trout age 1+ smolts following brush removal in River Arrigle in southeast Ireland. In contrast, Avery (2004) found that brush removal alone or combined with placement of cover logs provided little benefit to resident trout populations in Wisconsin streams. Some of the disparate results of these studies likely have to do with changes in light, organic inputs, and stream temperatures following removal of brush. These factors, unfortunately, were not examined in the studies we located on brush removal. Given the disparate results of these studies, drawing firm conclusions on the effectiveness of this technique is difficult; therefore, brush removal should be used with caution as habitat rehabilitation technique.

2.8.4 Habitat protection

While not typically considered a form of habitat rehabilitation, habitat protection is often the most cost-effective conservation method, as it is typically cheaper to protect habitat than to try and restore it later (NRC, 1992; Roni *et al.*, 2002). This concept is particularly important in undisturbed areas that are threatened by development or human uses. Habitat protection is often considered the first step when prioritizing habitat rehabilitation activities (Dominquez and Cederholm, 2000; Roni *et al.*, 2002). This often occurs in the form of land acquisitions (purchasing land), conservation easements (buying rights to development but not the land itself), or passing laws to protect areas from further development (Lucchetti *et al.*, 2005).

Although, the effectiveness of habitat protection strategies is generally apparent, monitoring and evaluation of their effectiveness is rarely conducted or published. Lucchetti *et al.* (2005) in a review of effectiveness of acquisitions and conservation easements in the United States, found no published evaluations of effectiveness. However, Lucchetti *et al.* (2005) provide guidelines for determining the level of monitoring and evaluation needed for various types of land acquisitions and conservation easements. As with other techniques, the lack of specific information on effectiveness does not mean they are ineffective. Clearly, the protection of high quality and important habitats is important throughout the world, and this underscores the need for thorough planning, careful implementation, and rigorous monitoring of these activities.

2.8.5 Conclusions – miscellaneous rehabilitation techniques

- Beaver reintroduction is an effective strategy for restoring floodplain and slow water habitats.
- Beaver removal may be effective for improving habitats for individual species, but can be detrimental to reestablishing natural processes and maintaining species and habitat diversity.
- Bank stabilization techniques can successfully reduce erosion, inhibit natural channel migration, and can be detrimental to fisheries resources.
- Similar to beaver removal, bank debrushing can improve conditions for specific fishes, but in general is detrimental to many fisheries resources and reestablishing natural processes.
- While not typically considered a rehabilitation technique, habitat protection should be considered prior to rehabilitation efforts.
- Few studies have evaluated habitat protection, but potential benefits are clear and it may be the most effective strategy in many areas.

3. Cost-effectiveness

3.1 REVIEW OF COST-EFFECTIVENES OF HABITAT REHABILITATION

Estimates of cost-effectiveness (i.e. increase in habitat area or fish numbers per unit cost) are even less frequently conducted than physical or biological evaluations of rehabilitation projects (Roni, 2005). This is in part because many projects are designed as experiments and accurate costs are not often recorded. Furthermore, many projects include in kind contributions, volunteers and other intangibles such as public education, which are difficult to quantify economically. Few studies we examined reported project costs and only a handful did any cost-benefit or economic analysis (e.g. Cederholm *et al.*,1997; House *et al.*,1989). However, the examination of the cost-effectiveness of fisheries habitat rehabilitation is a critical tool for assisting in the prioritization of restoration activities and allocation of resources for habitat rehabilitation and important part of the FAO Code of Conduct for Responsible Fisheries.

The available cost benefit or economic assessments of rehabilitation projects fall into three categories: 1) estimates of the benefit to the economy through estimates of expenditures by recreational fishers, 2) cost-benefit based on increased number of fish per dollar, and 3) a simple reporting of project costs. Below we review published cost-benefit analyses of freshwater rehabilitation projects in these three categories as well as provide estimates of average costs for different types of rehabilitation and discuss information needed for a proper cost-benefit or economic analysis.

Studies on the total benefit to the economy of aquatic habitat rehabilitation are limited. Of all the studies we examined only two examined total benefit to the economy and both of these were on large North American lakes. Hushak *et al.* (1999) examined two artificial reefs in a large lake and estimated that the total value of the reefs to the economy was US$276 000 per year versus a construction cost of $100 000 (1991 US$). Stockner and MacIsaac (1996) estimate that nutrient enrichment of British Columbia lakes had increased sockeye salmon returns to coastal British Columbia lakes worth approximately $16 million (1996 Canadian dollars) annually to the Canadian economy on an investment of about $17 million between 1976 and 1996. In a modeling effort on British Columbia sockeye salmon lakes, Guthrie and Peterman (1988) examined pulsed versus annual nutrient enrichment and found that the net economic benefit for annual nutrient enrichment was greater than for pulsed. These evaluations and modelling efforts demonstrate that lake enhancement can have large economic benefits.

The value to the economy of fish harvest has also been used but there is considerable debate about the economic value of various species and most work has focused on recreationally important species, most notably Atlantic salmon. The value of a single sport caught Atlantic salmon in 1991 ranged from 250 to 1 000 or more British pounds (Kennedy and Crozier, 1991). Thus if a project estimates the benefit in number of adult Atlantic salmon, a range of economic values can be applied to estimate benefits to the project. Atlantic salmon are unique because their harvest is highly regulated and access to a given section of stream is by fee, making calculations of their economic value more straightforward compared to other species.

Other studies have examined the cost versus the benefit in increased fish numbers and the value of those fish to the commercial fishery or the cost of producing fish through other means. House *et al.* (1989) examined the economics of 15 stream habitat enhancement projects along the Oregon coast (USA), and estimated that the economic benefits based on increased salmon and trout returns and harvest were more than three times the cost. Estimates such as House *et al.* (1989) do not contain the true economic value but the value of a landed fish (i.e. the price per kilogram paid to the fisher). O'Grady *et al.* (1991) estimated that the placement of rubble mats (boulders and cobble to create riffle or habitat diversity) in an Irish stream cost about 14 571 (IR) pounds, but provided an annual increase in juvenile and adult Atlantic salmon and brown trout worth 4 160 (IR) pounds based on the cost to rear and stock hatchery salmon and trout. In a more detailed evaluation of placement of instream structures and brush removal, O'Grady (1995) estimated that costs per returning adult salmon ranged from 0.4 to 8.3 (IR) pounds per additional returning adult Atlantic salmon. Based on their monitoring of nutrient additions in the Keogh River, British Columbia, Canada, Ashley and Slaney (1997) estimated the cost of producing an addition adult steelhead was between $11 and $20 Canadian depending upon the size of the stream.

Comparisons of the cost-effectiveness of different rehabilitation techniques are also rare and typically limited to structural manipulations. Examinations of Johnson and Lynch (1992) found that stake-bed structures used in lake enhancement in the midwestern US were not cost-effective in terms of fish response and angler use when compared with brush bundles or evergreen trees. O'Grady (1995) compared cost-effectiveness (cost per additional returning adult Atlantic salmon) of placement of instream structures, shrub-pruning (debrushing) and stocking of hatchery smolts and found shrub pruning and instream structures more cost-effective than stocking (1.38, 2.14, and 33.34 (IR) pounds, respectively). Cederholm *et al.* (1997) compared the cost per coho salmon for placing artificial structures versus simply falling trees in the stream and reported costs of US$14.82 versus $13.00 per smolt, respectively. However, he did not incorporate the time value of money over the 25-year period he examined, which would have greatly increased the cost of the artificial structure. This highlights a common problem in simple cost-benefit analysis conducted by fisheries scientists: ignoring basic financial and economic principles when conducting cost-benefit analysis (Plummer, 2005). Habitat protection is believed to be more cost-effective than structural manipulations or other habitat rehabilitation techniques, but similar to other techniques, little information exists on effectiveness of habitat protection measures (Lucchetti *et al.*, 2005).

Cost-benefit analyses can also be done using various "ecosystem services" such as water clarity, fish and wildlife habitat, allowable water uses, and naturalness (Holmes *et al.*, 2004). Using these values Holmes *et al.* (2004) estimated that riparian rehabilitation projects in the Little Tennessee River basin in North Carolina (USA) had a benefit to cost ratio ranging from approximately 4 to 16, suggesting that these efforts were economically feasible and that this approach might also be used to prioritize projects.

Given that existing economic evaluations of rehabilitation activities make a cost-benefit analysis or comparison of various project types virtually impossible, the next logical step would be to provide cost estimates for various types of rehabilitation. Estimating project cost would seem to be relatively straightforward; however, it is also difficult for most rehabilitation projects, as many have multiple government agencies and other entities that may not track their "in kind" contributions (staff time). Moreover, costs can vary considerably by site or region because of differences in site size, geology, biology, ownership, politics, permitting, and other factors. Thom *et al.* (1995) examined cost information on more than 90 wetland, instream, and estuarine rehabilitation or projects in the United States and found highly variable rates for basic construction activities (e.g. hauling logs, gravel). For example, gravel removal activity ranged in cost from US$327 to $3 239 dollars per ton and many other rehabilitation activities varied by two or three fold. Even within one type or area of stream rehabilitation costs can be highly variable.

The costs of similar projects per unit area will often vary by stream size. Ashley and Slaney (1997) reported costs of nutrient enrichment of British Columbia streams ranging from $180/km, $346/km, $571/km (Canadian dollars). In a review of Wyoming (USA) instream rehabilitation projects, Binns (1999) reported costs per kilometre in 1995 US dollars of $13 175, $16 780, $ 21 500, $22 000, and $43 000 for first, second, third, fourth, and fifth order streams respectively. Because thousands of dams built in the last hundred years are obsolete and in need of repair or removal in many developed countries and these projects are particularly costly, dam removal has recently received detailed economic analysis. Chisholm (1999) in a review of dam removal projects to date in the United States reported costs of removal ranging from $1500 to $3.2 million. Born *et al.* (1998) examined small dam removal projects that had occurred in 30 Wisconsin rivers and found that the estimated costs of repairing these dams were on average three times higher than the cost to remove them.

In an effort to provide a broader estimate of what various types of habitat rehabilitation might cost, we summarized the costs of a handful of studies reviewed in other sections of this document that reported costs (Table 15). These averaged from a US$3 500 per kilometre for decommissioning of a forest road to more than US$500 000 per kilometre for remeandering streams. The large variation in costs among and within project types emphasizes how site conditions and other factors can affect project costs. These numbers are probably much higher as many studies did not indicate specifically what year costs were incurred and we did not convert these to 2005 dollars. We also examined cost estimates for different categories of aquatic habitat rehabilitation projects implemented between 2000 and 2003 under the Pacific Coastal Salmon Recovery Fund, a $400 million plus initiative for stream rehabilitation in the northwestern United States and the National River Restoration Synthesis (NRRS) project (www.nrrss.umd.edu); Bernhardt *et al.*, 2005). Mean project costs within a category such as fish passage, ranged from a few thousand to several million dollars and averaged $1.2 million per project between 2001 and 2004 (Table 16). In contrast, costs of rehabilitation measures reported throughout the United States under the NRRS project were much higher. For example, average cost of instream habitat rehabilitation projects US$138 474 versus 365 545 for the Pacific Salmon Coastal Recovery Fund, and the NRRS project respectively. The differences in these two databases (Table 16) and from the literature (Table 15) emphasize differences in costs among and within regions and project types and the difficulty in obtaining accurate costs for a number of projects.

The time value of money (discount rate) and the annual benefits in dollars were examined in only a few studies – emphasizing the need for more detailed economic analysis. Plummer (2005) and Scarfe (1997) provide detailed information on how to conduct cost-benefit analysis. It is important to realize that when conducting cost-benefit analysis the metric chosen will have an effect on the cost-effectiveness of the project. For example, Plummer (2005) found differences in cost-effectiveness of several Pacific salmon habitat enhancement projects if increase in pool habitat or salmon production per unit cost were used.

3.2 CONCLUSIONS – COST-EFFECTIVENESS

In summary, we were not able to compare the cost and benefits of various habitat rehabilitation actions because actual costs are rarely reported for rehabilitation projects. Key items that need to be reported and considered for future projects so that cost benefit analysis will be possible for a single or multiple projects include:

- Full cost of project and year(s) incurred (including value of "in kind" contributions such as unpaid salaries etc.)
- Length or area of project
- Benefits in terms of physical habitat or fish numbers
- Discount rate when estimating multiyear benefits of projects
- Information on the economic value of fish, education, aesthetics, and other values.

TABLE 15

Average and range of costs of various rehabilitation activities reported in studies reviewed in previous sections of this document. We were unable to locate any grazing or riparian studies that provided a cost-benefit analysis. Holmes (1998), Ashley and Slaney (1997), Cederholm *et al.* (1997), and Lister and Finnegan (1997) are in 1995 US$ all others are as reported in reference. Unlike other techniques, nutrient enrichment needs to be repeated annually.

Rehabilitation technique	Number of sites examined	Mean expenditure US$	Mean US$/unit length or area	Range per unit length or area	Reference
Forest road decommissioning	1	51 650	3 500/km	NA	Harr and Nichols, 1993
Forest road removal	NA	NA	NA	3 100 to 155 400/ km	Switalski *et al*, 2004
Floodplain/off-channel ponds/sloughs	5 (1988 to 1994)	61 455	111 391/km	16 767 to 273 375/km	Cederholm *et al*, 1997
Floodplain/off-channel ponds/sloughs	9 (1990 to 1996)	85 730	7 81/m^2	1 9 to 13 12/m^2	Lister and Finnegan, 1997
Stream remeandering	5 (1988 to 1991)	262 000	584 919/km	348 000 to 1 000 000/km	Iversen *et al*, 1993
Stream remeandering	NA	194 600	138 575/km	NA	Nielsen, 1996
Stream remeandering	3 (1994 to 1995)	526 667	243 077/km	158 000 to 395 000/ km	Holmes, 1998
Stream nutrient enrichment*	3 (1993 to 1995)	6 561/yr	266/km/yr	131 to 416/km/ year	Ashley and Slaney, 1997
Instream structures	6 (1977 to 1994)	NA	49 937/km	13 122 to 162 296/ km	Cederholm *et al*, 1997
Instream structures	71 (1953 to 1998)	NA	24 243/km	2 957 to 188 630/km	Binns, 1999

NA: not available

Some of this information can and should be collected or reported while others will require detailed scientific as well as socioeconomic studies to determine the value of fisheries resources. This information can be used to assist in planning and prioritization of projects. The value of rehabilitated ecosystems to objectives other than fisheries (e.g. erosion control, climate amelioration, carbon sequestering) is extensive (Costanza *et al.*, 1997), but not considered here.

4. PRIORITIZING REHABILITATION ACTIVITIES

Our review of the effectiveness of different watershed rehabilitation techniques was undertaken in part to help determine which techniques might be most useful. Clearly many of the techniques we reviewed succeed or fail for similar reasons (Table 17). Most notably the actions that fail either ignore large scale-watershed processes such as sediment transport or basic fish needs such as water quality and adequate instream flow. Below we provide some guidance on planning and prioritizing rehabilitation actions based on our review. We emphasizes the need to restore basic processes and address factors limiting fish production and identify other factors to consider when more detailed information is available for prioritizing rehabilitation.

TABLE 16

Average cost and range of cost of selected habitat rehabilitation activities implemented under the Pacific Coastal Salmon Recovery Fund between 2000 and 2003 in the northwest United States and throughout the United States between 1990 and 2003 as reported by the National River Restoration Science Synthesis Project (www.nrrss.umd.edu; Berhardt *et al.*, 2005). The variation in costs regionally, nationally and among project types highlights the problems with accurate cost reporting and the range of costs among projects. Reported costs per project ranged from a few thousand dollars to several million and some projects covered more than one project type so costs may be inflated.

	Pacific Coastal Salmon Recovery Fund			National River Restoration Science Synthesis		
Rehabilitation project	Number of projects	Total Spent (US$ Millions)	Mean US$ per project	Number of projects	Total Spent (US$ Billions)	Mean US$ per project
Road and upland	420	57.8	137 588		NA	
Riparian habitat	524	62.1	118 455	11 881	3.23	271 664
Floodplain rehabilitation		NA		1 614	6.85	4 245 990
Fish passage	470	80.0	170 144	4 910	0.75	152 646
Dam removal		NA		799	1.16	1 451 862
Instream habitat	561	77.7	138 474	5 913	2.16	365 545
Land Acquisition	165	75.9	460 179	245	1.07	4 364 655
Water quality	256	36.6	142 928	12 076	5.38	445 543
Total number of projects	2 396	390.8	162 784	37 438	20.6	555 243

NA = not available

4.1 DETERMINING WHAT TO REHABILITATE

Two types of assessments or questions must be answered to identify necessary habitat rehabilitation actions and assist in planning and prioritization (Figure 12). The first group of assessments focuses on identifying disruptions to ecosystem function and the types of habitat rehabilitation necessary for ecosystem recovery. The second set of assessments concentrates on how humans have altered habitats and how those habitat changes have affected biota. Addressing these questions requires analysis of natural functioning of aquatic ecosystems (often including assessment of historical habitat types and abundance), as well as assessments of relationships between habitat and biota (Habersack, 2000; Beechie *et al.*, 2003a). These questions motivate assessments that identify where the biological integrity of ecosystems has been degraded and where specific ecosystem processes or functions are disrupted. In combination, these assessments provide a broad understanding of actions that are likely to improve the functioning of aquatic ecosystems, and form the basis of both regional and site-specific plans for ecosystem restoration.

Assessments of ecosystem functions and biological integrity can be separated into screening assessments that identify areas where ecosystem processes and functions are most impaired, and specific field inventories to diagnose causes of ecosystem impairment and opportunities for rehabilitation. Assessments that correlate landscape and land use characteristics with population attributes can indicate which habitat changes are most likely responsible for declines in specific organisms, and therefore which broad categories of rehabilitation actions are most likely to result in improved ecosystem functioning. Direct assessments of ecosystem processes that form aquatic habitats (e.g. barrier inventories, riparian condition inventories) identify causes of degradation, as well as rehabilitation actions that are required to recover ecosystem functions and biological integrity (Table 18).

TABLE 17

Summary of number of evaluations located on effectiveness of specific techniques, common parameters monitored and common limiting factors for each rehabilitation technique reviewed. Some studies examined more than one type of rehabilitation (i.e., placement of cover logs and spawning gravel).

Technique	Number of published evaluations of effectiveness	Parameter commonly examined	Common limiting factors
Roads/Sediment/Hydrology			
Sediment reduction (Resurface, stabilize slopes)	8	Fine and coarse sediment	Technique used
Hydrology (increase cross drains, replace culverts, etc)	4	Fine sediment, stability, peak flows	Number and type of structures used
Abandon or remove road	16	Landslides, sediment reduction	Replanting, removal of crossings, site prep
Riparian/Grazing			
Replanting and thinning	5	Plant growth, survival, WQ, inverts	Upstream conditions, protection from ungulates
Removal of exotic plants and misc. silviculture techniques	7	Species composition	Restoration of natural processes (flooding etc.)
Rest-rotation, other grazing systems	9	Plant growth, survival, WQ, invert, fish	Upstream conditions, monitoring of livestock levels, duration and season,
Fencing or removal of grazing	24	plant growth, survival, WQ, invert, fish	Upstream conditions, native herbivores, size of area protected, invasive species
Floodplain			
Connection of isolated habitat	12	Fish abundance, diversity	Access
Levee breaching setbacks	7	Complexity, sinuosity, connectivity, nutrients, fish & inverts	access, water quality and sediment
Channel reconstruction/ remeandering	8	Channel length, fish, inverts	water quality, sediment
Constructed habitats	7	Complexity, sinuosity, connectivity, nutrients, fish & inverts	Access, depth, complexity, exotic species
Dam Removal and Flood Flows			
Dam removal	14	Fish, sediment, WQ	WQ or toxic sediments, upstream conditions
Flood/High flows	12	Sediment, riparian, physical habitat, fish	Availability of sediment, exotic species, magnitude and frequency of flows
Instream structures			
Wood placement, log structures, rock structures	99	Habitat, fish, inverts	Upstream conditions, WQ
Cover structures and brush bundles	11	Habitat and fish	Upstream conditions, WQ
Gravel additions, spawning habitat	17	Juvenile and spawner abundance	Upstream conditions and processes
Lake Habitat Enhancement			
Wood or other cover structures	37	Fish use and survival	Material used, depth, temperature, species
Spawning reefs or habitat	9	Fish use and survival	Depth, species, location near rearing habitats
Nutrient enrichment			
Addition of organic or inorganic nutrients	17 streams 36 lakes	Nutrients, primary productivity, fish growth, survival, abundance	Initial nutrient status, species composition, physical limnology, limiting factors

FIGURE 12

Schematic diagram of linkages among landscape processes, land use, habitat change, and biological responses (adapted from Beechie *et al.*, 2003b). Assessing disrupted ecosystem functions and processes identifies causes of habitat change that result in diminished biological integrity and declines in fish populations.

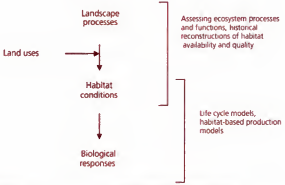

Landscape processes
Land uses
Habitat conditions
Biological responses
Assessing ecosystem processes and functions, historical reconstructions of habitat availability and quality
Life cycle models, habitat-based production models

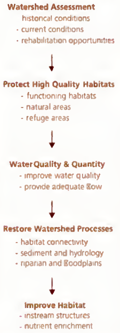
TABLE18

Examples of landscape processes and functions that should be addressed in planning watershed and river restoration.

Distributed watershed processes
Hydrology – peak flows, low flows, and channel forming flows
Sediment supply- erosion and delivery of sediment
Delivery of contaminants, nutrients
Reach-level processes
Riparian functions – shade, litter delivery, wood delivery, resistance to bank erosion
Channel and floodplain interactions – channel migration, flooding
Habitat connectivity
Blockages to upstream migration (e.g. dams)
Blockages to lateral migration (e.g. levees)

Analysis of habitat change and influences on biota help predict fish and biota response to habitat changes. Common methods include assessment of habitat change to estimate or model changes in fish populations or other biota (e.g. Beechie *et al.*,1994, Kemp *et al.*,1999, Muhar *et al.*,2000), and correlation analyses that relate landscape and land use characteristics to fish populations and communities without directly quantifying changes to habitat (e.g. Feist *et al.*, 2003, Steel *et al.*, 2004, Filipe *et al.*, 2004). It should be noted that these type of analyses do not directly identify causes of habitat degradation or specific rehabilitation actions. Rather they provide a means of predicting ecosystem responses to rehabilitation actions (e.g. habitat-based estimates of potential population size for specific organisms), insights into potential changes in species diversity or life history diversity, and a means of identifying which species or populations are most constrained by habitat loss and therefore may be most difficult to recover (Beechie *et al.*, 2003b, Filipe *et al.*, 2004, McHugh *et al.*, 2004). Thus, when combined with impaired processes and potential rehabilitation actions developed from the assessments of ecosystem functions and biological integrity they assist in the planning and prioritization of rehabilitation.

FIGURE 13
Strategy for prioritizing watershed rehabilitation actions. First step (box) is to conduct assessments necessary to determine potential opportunities.



4.2 PRIORITIZING ACTIONS

Once the previously described assessments have been completed and a list of potential actions has been established there are several approaches for prioritizing rehabilitation actions depending on the level of information available. Based on our understanding of watershed processes, basic needs of aquatic biota, and our review of effectives of techniques, we recommend an interim approach for setting rehabilitation priorities. This begins first with protecting high quality habitats and providing adequate water quality and quantity and is then followed either sequentially or concurrently by restoring connectivity of habitats, restoring processes that create and maintain habitat, and finally, if necessary, instream habitat improvements (Figure 13). We begin this section by discussing this interim approach and then outlining other approaches for prioritizing rehabilitation.

Habitat protection is typically not seen as rehabilitation. However, given the loss of habitat and the continued degradation from human activities, protecting high quality functioning habitats should be part of the rehabilitation planning process (NRC, 1992; Roni *et al.*, 2002; Lucchetti *et al.*, 2005). As discussed in previous sections, it is also considered more cost-effective and reliable than trying to rehabilitate habitat once it has been degraded.

Adequate water quality and quantity are fundamental to the success of any rehabilitation action or suite of actions. Our review of published data and anecdotal evidence suggests that inadequate water quality and quantity have prevented many expensive rehabilitation projects from achieving successful increases in fish production. In some cases adequate water quality can be achieved through riparian protection or replanting, such as in the case of agricultural runoff (Sovell *et al.*, 2000; Meals and Hopkins, 2002; Parkyn *et al.*, 2003). However, pollutants often originate from both point and nonpoint sources including runoff from roads, treated or untreated sewage effluent, or runoff from agricultural lands and will require much more complicated measures to improve water quality. Water quantity is typically addressed by limiting the timing and volume of water withdrawals or water use. Lengthy political, legal, or legislative acts may be needed to restore water quantity and quality. While both water quality and quantity can be difficult to address, they are critical to the success of other rehabilitation actions.

Assuming problems with water quality and quantity have been or are being addressed; we suggest that restoring connectivity of habitats is the next factor to consider. Reconnection of isolated habitats relies on existing habitat and often produces both rapid and long-term results. This includes both restoring longitudinal connectivity such as removal or providing passage at manmade barriers as well lateral connectivity such as reconnecting isolated floodplain habitats and floodplains. The techniques discussed in the section on floodplain connectivity and the culvert/fish passage discussion in the roads section have all shown rapid and promising results, assuming water quality and instream flow issues have been addressed.

Setting up a watershed for long-term recovery will require the restoration of natural processes such as delivery of sediment, organic material, nutrients, and other processes. Thus the next logical step and the factors that most often limit the success of structural manipulations of instream habitat are the restoration of basic hydrologic, geologic, and riparian processes.

While considerable effort and focus is placed on structural manipulations of habitat such as placement of boulders and wood, and many have shown success at increasing local fish numbers, their sucess is often determined by the previously discussed factors (water quality and quantity, habitat connectivity, and watershed processes). Thus we recommend this category of techniques be implemented following or in conjunction with improvement of those factors.

The methods discussed above and in Figure 13 represent a starting point for restoring a watershed, given fairly broad goals. In many cases, rehabilitation actions may be focused on one or two species. In watersheds where information exists on factors limiting specific species, a more detailed prioritization of actions is possible. Roni *et al.* (2002) provided a general strategy for prioritizing actions for Pacific salmon (Figure 14). Beechie and Bolton (1999) using a relatively coarse level of resolution provided an example of how priorities for habitat rehabilitation might differ between two salmon species with different life histories and habitat requirements (Figure 15).

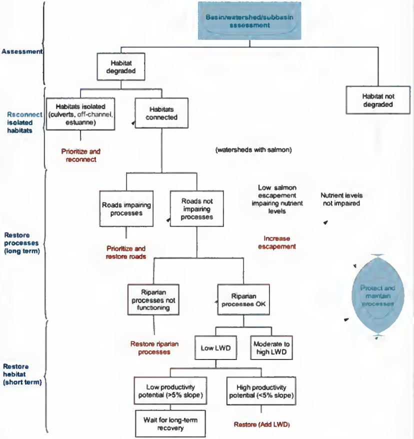
It is critical to bear in mind that the prioritization of actions such as outlined in Figure 15 does not alter the types of actions that are needed to restore ecosystems. Ecosystems restoration will include a wide range of recovery actions affecting the entire life cycles of multiple species. Where a single species is a primary concern, altering the sequence of those actions for rapid recovery of that species might be prudent. The limitation of a single species approach is demonstrated in western North America, where efforts initially focused on rehabilitating habitat for one species of Pacific salmon, but quickly had to change focus when additional species became threatened.

Alternative strategies for prioritizing rehabilitation that incorporate economic, ecologic, and biologic factors have been proposed or used, especially where there are multiple species of concern and prioritizing actions based on the needs of individual species will lead to conflicting priorities. For example, Sedell *et al.* (1990), Wasserman *et al.* (1995), Beechie *et al.* (1996), Frissell (1993), Frissell and Bayles (1996), and others have outlined rehabilitation strategies that focus on providing refugia and protecting high quality habitats. Beechie *et al.* (1996) outlined a prioritization strategy that focused on providing refugia for a depressed steelhead trout population in Deer Creek, Washington (USA). Other strategies might prioritize actions on potential increase in fish numbers, cost, cost per fish, aquatic diversity, assuring for metapopulations structure or diversity, or scoring based on a suite of these and other factors (e.g. Beechie *et al.*, 1996, Frissell and Bayles, 1996; Doyle, 1997). The "Leitbild" (target vision) concept developed in Austria, which is similar to process based restoration, but includes setting numerous goals and priorities with public input, is widely accepted and consistent with European Water Framework Directive (Muhar *et al.*, 2000; Jungwirth *et al.*, 2002). These various strategies incorporate management goals beyond simply restoring watershed or ecosystem processes and habitat. Where at least one species appears to be at high risk of extinction, the refugia approach may be most appropriate to make sure that individual populations are preserved first. By contrast, watersheds with relatively stable populations might embark on a longer term, process-based approach to ecosystem recovery previously outlined (Figure 13 and 14).

The sequencing of rehabilitation actions under different prioritization strategies will vary. We demonstrate how priorities might differ based on different restoration prioritization schemes by running alternative scenarios. First, we developed a

FIGURE 14

Flow chart depicting hierarchical strategy for prioritizing specific rehabilitation activities (modified from Roni *et al.*, 2002). Shaded boxes indicate where rehabilitation actions should take place. Addition of organic or inorganic nutrients may be appropriate at various stages following reconnection of isolated habitats. LWD = large woody debris.


Basin/watershed/subbasin assessment
Assessment
Habitat degraded
Habitat not degraded
Reconnect isolated habitats
Habitats isolated (culverts, off-channel, estuarine)
Habitats connected
Prioritize and reconnect
(watersheds with salmon)
Roads impairing processes
Roads not impairing processes
Low salmon escapement impairing nutrient levels
Nutrient levels not impaired
Restore processes (long term)
Prioritize and restore roads
Increase escapement
Riparian processes not functioning
Riparian processes OK
Protect and maintain processes
Restore riparian processes
Low LWD
Moderate to high LWD
Restore habitat (short term)
Low productivity potential (>5% slope)
High productivity potential (<5% slope)
Wait for long-term recovery
Restore (Add LWD)

hypothetical list of potential rehabilitation actions along with detailed information on their cost, length and area restored, whether they provide refugia for endangered species, potential increase in fish numbers, and cost per fish (Table 19). Second, we ranked rehabilitation actions using different prioritization schemes mentioned above (Table 20). This analysis demonstrated that if rehabilitation actions were prioritized based on Figure 13 or 14, impassible culverts and reconnection of habitats would occur first, followed by road, riparian, and LWD placement. If actions were prioritized by whether they were refugia for an endangered species, instream flow and LWD placement would be first, simply because they are in a high priority area. Similarly, different cost, cost/fish, and total fish production all produced slightly different prioritization scenarios. This simple example illustrates how priorities might differ based on the method, information used, and management objectives.

4.3 CONCLUSIONS FOR PRIORITIZATION

The appropriate method for prioritizing rehabilitation activities within a watershed will depend on numerous factors. However, our review of rehabilitation techniques suggests that an initial approach such as Figure 13 or the Lietbilt approach described by Jungwirth *et al.* (2002) may help lead to more successful rehabilitation projects. This approach can then be modified as more detailed information watershed conditions, potential projects and other factors become available.

FIGURE 15

Sequence and prioritization of habitat rehabilitation based on species of interest. Modified from Beechie and Bolton (1999).

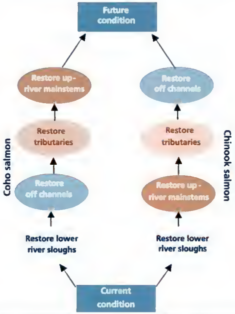


TABLE 19

Hypothetical example of list of potential rehabilitation actions within a watershed. Refugia are based on Beechie *et al.* (1996) and include 1 = refugia (areas where recovery is relatively predictable), 2 = key habitat areas that provide for the largest long-term recovery of species of interest, but are sensitive to disturbance and more difficult to restore, and 3 = habitat areas expected to provide the smallest gain for species of interest. Coho and Chinook smolts represent expected annual increase in smolt production. All numbers are fictitious but reasonable and for demonstration purposes only. Modified from Beechie *et al.* (2003b).

Site ID	Action	Refugia	Km treated	M treated	coho smolts	Chinook smolts	Cost in US$	$ per coho
A	LWD placement	2	3	15 000	3 750	750	50 000	13.3
B	LWD placement	1	2	10 000	2 500	500	32 000	12.8
C	LWD placement	1	2	14 000	3 500	700	35 000	10
D	Fencing	1	5	60 000	6 000	3 000	20 000	3.3
E	Increase flows	2	20	200 000	20 000	10 000	500 000	25
F	Reconnect tidal channels	3	1	100 000	10 000	50 000	350 000	35
G	Create new estuarine slough	3	2	200 000	20 000	100 000	750 000	37.5
H	Culvert replacement/ fish passage	2	3	15 000	3 750	0	150 000	40
I	Road decommissioning	1	20	200 000	20 000	10 000	1 500 000	75
J	Road resurfacing/ sediment reduction	2	10	50 000	5 000	2 500	750 000	150
K	Reconnect isolated oxbow slough	3	4	800 000	400 000	40 000	75 000	0.19

TABLE 20

Example of different order of priorities based on use of different prioritization schemes using information presented in Table 19. Roni *et al.*, (2002) and refugia methods of prioritization do not distinguish between projects of the same type. Hence, there are only four levels and three levels, for the two methods, respectively. Modified from Beechie *et al.* (2003b).

Site ID	Potential Rehabilitation Action	Roni	Refugia	Total Cost	Cost/coho	Total Fish
A	LWD placement	3	2	4	5	4
B	LWD placement	3	1	2	4	1
C	LWD placement	3	1	3	3	3
D	Fencing/cattle exclusion	3	1	1	2	6
E	Instream flows/Purchase water rights	1	2	8	6	8
F	Reconnect estuarine tidal channel (dike removal)	1	3	7	7	9
G	Excavate new estuarine slough	4	3	10	8	10
H	Culvert replacement/fish passage	1	2	6	9	2
I	Road decommissioning	2	1	11	10	7
J	Road resurfacing/sediment reduction	2	2	9	11	5
K	Reconnect isolated oxbow slough	1	3	5	1	11

5. MONITORING AND EVALUATION

Throughout or review of the effectiveness of different habitat rehabilitation techniques we have emphasized the need for better monitoring and evaluation. Our understanding of the effectiveness of different habitat rehabilitation is limited because monitoring has often not been adequately replicated spatially (i.e. number of sites) and temporally (i.e., too short) and often designed as an afterthought. Designing appropriate monitoring and evaluation programmes for stream rehabilitation will differ by project type as well as by region, geomorphology, scale, and a host of other factors. However, there are several basic steps that must be taken to design an effective monitoring and evaluation programme that will allow us to learn more about the rehabilitation techniques. In an effort to provide clear guidance on monitoring and evaluation, we provide a brief overview of steps and considerations for developing a rigorous monitoring and evaluation programme for single or multiple projects and at fine (habitat or reach) or coarse scales (watershed) and provide key references were more detailed information can be obtained. We draw heavily from Roni (2005) and refer the reader to this reference for an in-depth treatment of the concepts we discuss in this section.

A well-designed monitoring and evaluation programme is a critical component of any resource management, conservation, or rehabilitation activity. It can also help reduce the cost and increase the benefits of future rehabilitation in part by minimizing failures (Lewandowski *et al.*, 2002). The development of a monitoring programme is best done as an integral part of the design phase of rehabilitation. Many previous studies have been of limited usefulness because they were not designed and implemented as part of the initial rehabilitation project. The objectives of individual rehabilitation programmes and projects vary, as do the objectives of monitoring programmes. Numerous decisions that need to be made in designing a monitoring programme are often interrelated with those that need to be made in developing a rehabilitation project. Thus the two should occur concurrently well before construction of the project occurs. That is not to say that retrospective studies of past rehabilitation activities are not without utility, but that most questions or hypotheses will require collection of data before and after rehabilitation.

TABLE 21

Definitions of monitoring types (adapted from MacDonald *et al.*,1991 and Roni, 2005) and examples of what might be monitored for a wood placement project targeting fish. Effectiveness and validation monitoring are typically the types used to habitat evaluate rehabilitation actions.

Monitoring types (other names)	Description (hypotheses)	Examples
Baseline	Characterizes the existing biota, chemical, or physical conditions for planning or future comparisons	Fish presence, absence, or distribution
Status	Characterizes the condition (spatial variability) of physical or biological attributes across a given area	Abundance of fish at time x in a watershed
Trend	Determines changes in biota or conditions over time	Spawner surveys and temporal trends in abundance
Implementation (administrative, compliance)	Determines whether project was implemented as planned	Did contractor place number and size of logs as described in plan?
Effectiveness	Determines whether actions had desired effects on watershed, physical processes, or habitat	Did pool area increase?
Validation (research, sometimes considered part of effectiveness)	Evaluates whether the hypothesized cause and effect relationship between rehabilitation action and response (physical or biological) were correct	Did change in pool area lead to desired change in fish or biota abundance?

5.1 DEFINITION OF MONITORING AND EVALUATION

Before discussing steps for monitoring and evaluation it is important that we define what we mean by monitoring, as there are several types. As with the terminology of restoration, there is much confusion about monitoring terminology or types. Monitoring is technically defined as systematically checking or scrutinizing something for the purpose of collecting specified categories of data. In ecology it generally refers to sampling something in an effort to detect a change in a physical, chemical, or biological parameter. The common types of monitoring used to examine changes in aquatic habitat and biota include: baseline, trend, implementation, effectiveness, and validation monitoring (Table 21; MacDonald *et al.*, 1991; Roni, 2005). Determining whether a rehabilitation project was implemented correctly (implementation or compliance monitoring) is an important part of understanding why it may or may not have achieved goals and objectives. Implementation monitoring is relatively straightforward, involves quality assurance and project construction management, and may be as simple as a yes-no checklist (Kershner, 1997). Effectiveness and validation monitoring, which typically focus on determining whether an action had the desired physical and biological effects (Table 21), are often much more complex, more difficult, and longer term than implementation monitoring. They are also the type of monitoring we use to evaluate rehabilitation actions and the focus of our discussion. Other types of monitoring (status and trend) may also help plan and inform evaluation of rehabilitation actions.

5.2 STEPS FOR DEVELOPING MONITORING PROGRAMMES

Regardless of the type, number, and scale of aquatic rehabilitation actions, there are several logical steps that should be taken when designing any monitoring and evaluation programme. These include establishing project goals and objectives, defining clear hypotheses, selecting the monitoring design, selecting monitoring parameters, spatial and temporal replication, selecting a sampling scheme for collecting parameters, implementing the programme, and finally, analyzing and communicating results (Figure 16). Many of these steps are interrelated and some steps could occur simultaneously or in a different order than presented here. For example, monitoring design depends on hypotheses and spatial scale, just as the number of sites or years to monitor depends in part on the parameters selected. The first steps are critical for designing an effective monitoring and evaluation programme and we focus our discussion on these.

Determining the objectives of the project and defining key questions and hypotheses are the critical first steps in developing a monitoring programme. Defining the key questions will depend on the overall project objectives. Evaluation of rehabilitation actions can be broken down into four major questions based on scale (e.g. site, reach, watershed) and desired level of inference (number of projects). These include evaluations of single or multiple reach-level projects and single watershed or multiple watershed-level projects (Table 22). For example, if one is interested in whether an individual rehabilitation action affects local conditions or abundance (reach scale), the key question would be: What is the effect of rehabilitation project x on local physical and biological conditions? In contrast, if one is interested in whether a suite of different project types has a cumulative effect at the watershed scale, then the key question would be: What is the cumulative effect of all rehabilitation actions within the watershed on physical habitat and populations of fish or other biota? While some actions such as riparian plantings or instream wood placement can cover multiple adjacent reaches or occur in patches throughout a geomorphically distinct reach, the initial question is still whether one is interested in examining local (site or reach scale) or watershed-level effects on physical habitat and biota.

Determining the scale of influence for physical habitat responses requires distinguishing between habitat unit, reach, and watershed-scale effects (Frissell and Ralph, 1998; Roni *et al.*, 2003). However, for fishes and other mobile organisms,

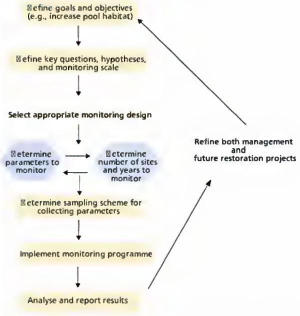
TABLE 22

Overarching hypotheses for monitoring aquatic rehabilitation divided by scale and number of projects of interest (from Roni, 2005). Most appropriate study designs are listed in parentheses. BA = before-after study design, BACI = before-after control-impact, and EPT = extensive post-treatment design. Extensive design refers to a design that is spatially replicated (many study sites, reaches, or watersheds).

	Spatial Scale	
Number of projects	Reach/local	Watershed/population
Single project	Does single project effect habitat conditions or biota abundance? (BA or BACI	Does an individual project affect watershed conditions or biota populations? (BA or BACI)
Multiple projects	Do projects of this type affect local habitat conditions or biota abundance? (EPT or replicated BA or BACI)	A. What are the effects of a suite of different projects on watershed conditions or biota populations? (BA or BACI) B. What is the effect of projects of type x on watershed conditions or biota populations? (BA or BACI)

FIGURE 1fl

Key steps for developing a monitoring and evaluation programme for rehabilitation actions. Modified from Roni (2005).


Define goals and objectives (e.g., increase pool habitat)
Define key questions, hypotheses, and monitoring scale
Select appropriate monitoring design
Determine parameters to monitor
Determine number of sites and years to monitor
Determine sampling scheme for collecting parameters
Implement monitoring programme
Analyse and report results
Refine both management and future restoration projects

determining the appropriate scale requires differentiating between changes in local abundance and changes in population parameters at a watershed or larger scale. Most research on habitat and biota, both for rehabilitation and other ecological studies, has focused on reach scale or individual habitat units. This information is important, but uncertainty about movement, survival, and population dynamics of biota prevent these reach-scale studies from addressing watershed or population-level questions. Studies designed to assess watershed or population-level effects can provide valuable information but also face multiple challenges (e.g. upstream-downstream trends, sampling logistics; Conquest 2000; Downes *et al.*, 2002).

From the key questions and specific hypotheses will flow the other important decisions including appropriate monitoring design, duration and scale of monitoring, sampling protocols, etc. The most difficult part and the biggest shortcoming of many rehabilitation evaluation programmes is the study design. As noted in our review of riparian rehabilitation, lack of preproject data, adequate treatments and controls, reference sites, and various management factors have limited the ability of many studies to determine the effects of rehabilitation actions. There are many potential study designs for monitoring single or multiple rehabilitation actions. None is ideal for all situations and each has its own strengths and weaknesses. Hicks *et al.* (1991) distilled these possibilities down to a handful of experimental designs based on whether data are collected before and after treatment (before-after, or post-treatment designs) and whether they are spatially replicated or involved single or multiple sites (intensive or extensive). They also described the pros and cons of each approach (Table 23). Many variations of these basic study designs have been used or proposed in monitoring of land use, pollution, and habitat alterations (e.g. Johnson and Heifetz, 1985; Walters *et al.*,1988; Bryant, 1995) and can easily be modified for use in evaluating rehabilitation actions. However, most of these modifications can be classified as either before-after or post-treatment study designs. The first include collection of data before and after implementation of the rehabilitation project often with a control reach or watershed (before-after control-impact or BACI design) and the later are retrospective studies implemented after rehabilitation and rely on comparing treated areas to suitable control (same but no treatment) or reference (ideal or natural conditions) areas. The many strengths and weaknesses of different designs are thoroughly reviewed in Hicks *et al.* (1991), Downes *et al.* (2002), and Roni (2005) (Table 23). No one design is correct for all situations—the key questions and hypotheses will help determine the most appropriate design.

Determining which metrics and parameters to monitor and measure logically follows defining goals and objectives, key questions and hypotheses, definition of scale, and selection of study design. Selecting parameters also goes hand in hand with spatial and temporal replication and sampling schemes discussed below. Parameters and metrics should not be selected arbitrarily or simply because they were used in other studies. Monitoring parameters should be relevant to the questions asked, strongly associated with the rehabilitation action, ecologically and socially significant, and efficient to measure (Downes *et al.*, 2002; Bauer and Ralph, 2001; Kurtz *et al.*,2001). For example, monitoring of riparian rehabilitation will likely be focused on indicators of plant growth and diversity as well as some channel features, while instream habitats improvement may focus on instream habitat features and changes in fish numbers or diversity. Moreover, to be useful the parameter must change in a measurable way in response to treatment, be directly related to resource of concern, and have limited variability and not likely be confounded by temporal or spatial factors (Conquest and Ralph, 1998).

TABLE 23

Summary of advantages and disadvantages of the major study designs used to evaluating stream or watershed rehabilitation or habitat alteration (modified from Roni, 2005). Intensive study design generally includes sampling at one or two study sites or streams, extensive at multiple study sites, streams, or watershed. Years of monitoring needed to detect a fish response are general estimates based on juvenile salmonid studies and extensive study designs assume more than 10 sites are sampled (space for time substitution) thus fewer years of monitoring are needed.

	Study Designs				
	Before and After			Post-treatment	
Attribute (pros and cons)	Intensive	Extensive	BACI	Intensive	Extensive
Includes collection of preproject data	yes	yes	yes	no	no
Ability to assess interannual variation	yes	yes	yes	yes	no
Ability to detect short-term response	yes	yes	yes	no	yes
Ability to detect long-term response	yes	no	yes	yes	yes
Appropriate scale (WA = watershed, R=Reach)	R/WA	R/WA	R/WA	R	R/WA
Ability to assess interaction of physical setting and treatment effects	low	high	low	low	high
Applicability of results	limited	broad	limited	limited	broad
Potential bias due to small number of sites	yes	no	yes	yes	no
Assume treatment and controls are similar before treatment	NA	NA	no	yes	yes
Results influenced by climate, etc	yes	yes	yes	yes	no
Years of monitoring needed to detect a fish response	10+	1-3	10+	5+	1-3

NA = not applicable

The appropriate parameters to monitor will differ by types of rehabilitation as well as specific hypothesis. The choice of a parameter should in part be based on the different sources of spatial and temporal variability associated with that parameter. Both observation error and natural variability of a quantity will reduce the precision with which the mean of the quantity is estimated. For example, electrofishing and snorkeling are both used to estimate juvenile fish densities in small streams. While electrofishing may have a smaller observation error, it is more time consuming and thus leads to fewer surveyed habitat units. If the variability of fish is high between units then the marginal reduction in observation error may have a relatively small effect on the precision of the mean density estimate when compared with the increase in precision from snorkeling more units. Moreover, temporal variation within sites and across sites can affect the usefulness of an indicator or parameter for detecting local and regional trends in biota or habitat (Larsen *et al.*, 2001). It is important to consider these different types of error when selecting monitoring parameters.

Numerous publications discuss different parameters to measure, the strength and weaknesses. Many regional protocols exist for different parameters. Parameters typically address watershed processes or physical, chemical, and biological changes. We summarize common parameters in each of these categories in Table 24 and attempt to link them to basic watershed processes as outlined in Figure 1. The reader should consult regional protocols for more information on which might be most useful in their region.

Determining the spatial and temporal replication needed to detect changes following rehabilitation can and should be established prior to monitoring. This will also help determine whether the initial parameters selected will be useful in detecting change to the rehabilitation action in questions. This can be done using relatively straightforward power analysis found in statistical software packages and statistical texts. Similarly sampling schemes for collecting data within a given study area are covered in similar texts (e.g. simple random, systematic, stratified random, multistage, double sampling, Line transect).

Once the monitoring programme has been designed and implemented, the results obviously need to be written up and published. While this seems intuitive, many studies on habitat rehabilitation have only been published as grey literature. Moreover, the published literature is likely biased towards projects that showed an improvement following rehabilitation. Rehabilitation actions are experiments and reporting both positive and negative findings are critical for improving our understanding of the effectiveness of different measures, spending limited rehabilitation funds wisely, and restoring aquatic habitats and ecosystems.

5.3 CONCLUSIONS FOR MONITORING AND EVALUATION

Monitoring and evaluations is critical to understanding rehabilitation actions and spending future funds wisely. Key factors to consider when developing a monitoring and evaluation programme for rehabilitation include:

- Establishing hypotheses, choosing an appropriate study design, and selecting sensitive parameters linked to the hypotheses.
- The lack of published evaluations of habitat rehabilitation emphasize the need for better reporting and publishing both successful and unsuccessful projects.
- Design of monitoring is best done as part of project design, not as an afterthought.
- The failure to detect significant changes in watershed processes, physical habitat, or biota has often been because of poorly designed monitoring and not following the steps defined above.

TABLE 24

Common parameters utilized to evaluate rehabilitation projects

Rehabilitation technique	Watershed/riverine processes	Physical habitat	Water/nutrients	Biota
Roads/sediment/ hydrology	Mass wasting (landslide) rate and volume, fine sediment and coarse sediment delivery and storage, surface erosion, hydrology (discharge), connectivity of roads with stream channel, sediment storage and transport, bed scour and fill	Channel cross sections and long profile, channel width, channel units (habitats), residual pool depth, fine sediment, substrate size and composition	Turbidity, nutrients, water chemistry	Macroinvertebrate diversity and abundance, fish abundance, diversity, and survival
Riparian	Sediment and nutrient transport and retention, channel aggradation and migration, wood transport and retention, changes in hydrology and groundwater levels, vegetation succession and composition	Channel geometry, large woody debris, fine sediment, bank stability, soil conditions, percent cover, habitat quality (pool depth), shade and canopy cover	Temperature, nutrients, water and soil chemistry	Vegetation species composition, diversity, growth, survival, biomass, root density; vertebrate and invertebrates measurements may also be appropriate for some projects
Floodplain	Rate of channel migration, Sediment transport and storage, Wood transport and retention, hydrology, nutrient transport and retention, riparian species succession and composition	Channel geometry, channel pattern, length, density; habitat units	Temperature, nutrients, water and soil chemistry	Juvenile and adult fish diversity abundance, survival, movement; macroinvertebrate diversity, abundance, periphyton and aquatic macrophyte growth, species composition, biomass
Instream structures	NA	Habitat units, channel morphology, large woody debris, cover, substrate, cross sections, long profile	Temperature	Juvenile and adult fish diversity abundance, survival, movement; macroinvertebrate diversity, abundance, periphyton and aquatic macrophyte growth, species composition, biomass
In-lake structures	NA	NA	NA	Juvenile and adult fish species abundance, movement
Nutrient enrichment	Nutrient retention and uptake including stable isotope analysis of biota	NA	Chemistry and nutrients	Periphyton, primary production and chlorophyll a, macroinvertebrate and fish growth, abundance and biomass
Acquisitions and conservation easements (habitat protection)	Hydrology, connectivity of habitats, channel migration, sediment transport	Landowner and human use, see also list under floodplain rehabilitation	Temperature, chemistry, nutrients	Species composition and richness, invasive species presence or absence of rare or sensitive species, behaviour of biota (reproduction, rearing, refuge, migration), abundance, survival, and growth of key species

6. OVERALL SUMMARY

The intent of this document has been to summarize what is known about the effectiveness of habitat improvement techniques for inland fisheries and provide guidance for rehabilitation of freshwater ecosystems throughout the world. Limited information exists on the effectiveness of most techniques in restoring natural watershed processes or increasing fish and other biota abundance. However, techniques such as reconnection of isolated habitats, rehabilitation of floodplains, and placement of instream structures have proven to be effective at improving habitat and increasing local fish abundance. Techniques that restore processes such as riparian rehabilitation, sediment reduction methods (road improvements), and restoration of floods or high flows in regulated rivers also show promise. Other techniques commonly referred to as rehabilitation such as bank protection, beaver removal, and bank debrushing can produce positive effects, but more often produce negative impacts on biota or disrupt natural processes. Our review demonstrates that three key areas are lacking in rehabilitation activities: 1) adequate assessment of historic conditions, impaired ecosystem processes, and factors limiting biotic production; 2) understanding upstream or watershed-scale factors that may influence effectiveness of reach or localized rehabilitation techniques; and 3) well designed and funded monitoring and evaluation studies. These are generally the factors that consistently limit the ability of published studies to determine the success of a given technique at improving habitat conditions or fisheries resources.

Given the uncertainty in results of many habitat rehabilitation techniques and the continued threats to aquatic ecosystems, habitat protection should also be considered a part of habitat rehabilitation activities. Moreover, it is generally more cost-effective to protect existing high quality habitats and functioning ecosystems than it is to try to rehabilitate them once they have been degraded. High quality, unimpacted areas often provide important refuges for endemic species, critical information on the structure and function of natural habitats, and can serve as a reference or benchmark for rehabilitation of degraded habitats and ecosystems.

To assure that rehabilitation actions undertaken in the future are effective we suggest that the following steps be followed when planning, prioritizing, and implementing aquatic rehabilitation: 1) conduct assessments to determine historic and current conditions, disrupted processes, and rehabilitation opportunities, 2) protect high quality habitats and address water quality and quantity issues, 3) restore watershed processes and connectivity of habitats, then 4) improve reach or localized habitat conditions. Completing the first three steps will assure that habitat enhancement methods such as placement of instream structures or other in-channel activities will be successful and have the desired effects.

Commensurate with planning and implementing rehabilitation activities is the need for well designed monitoring with adequate spatial (number of sites) and temporal replication (length of monitoring). Despite the fact that we located and reviewed 334 published evaluations of rehabilitation actions, it is difficult to draw firm conclusions from many of these studies or provide concrete recommendations on which actions are successful under which conditions or for specific species. Thus well-designed monitoring and evaluation that includes specific hypotheses, adequate study design and replication in space in time (number of sites and duration of monitoring), and selection of sensitive parameters is desperately needed (Figure 16). This needs to be instituted during planning and design of the rehabilitation activities, rather than an afterthought. In addition to scientific monitoring and evaluation, detailed costs should be reported for each project. Few studies have done a cost-benefit analysis to help plan rehabilitation in part because of the lack of accurate cost data provided for various projects. Thus accurate project cost should be reported as part of comprehensive monitoring and evaluation to assist with future cost-benefit and economic analysis. Just as reporting scientific results will further restoration science, reporting cost-benefit information will assist in the socioeconomic management of habitat rehabilitation actions regionally, nationally and internationally.

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Glossary

Acquisition	The purchasing of a piece of land for the protection or restoration of plants and animals. Commonly called fee simple acquisition by real estate and land trust practitioners, it is the result of purchasing all rights an existing owner may have to the land, including that of "quiet enjoyment" in perpetuity. Compare *easement.*
Anadromous	An organism that migrates from the sea to freshwater for reproduction.
Anthropogenic	Caused or produced by human action.
Bankfull width	Channel width between the tops of banks on either side of a stream; tops of banks are the points at which water overflows its channel at bankfull discharge. Contrast *wetted width.*
Baseline monitoring	Characterizing existing biota, chemical or physical conditions for planning or future comparisons. Compare *status, trend, implementation, effectiveness, and validation monitoring.*
Basin	See *watershed.*
Benthic	Of, related to, or living in the soil–water interface of a lake or stream.
Biofilm	Community of microorganisms attached to a solid surface.
Biotic integrity	The capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitat of the regions.
Biomass	Total amount of all living organisms in a biological community, as in a unit area or weight or volume of habitat.
Biota	Flora and fauna of a region.
Brush bundles	Groups of trees and branches placed into a lake or stream to create habitat and cover for fish. Often used in conjunction with other habitat rehabilitation techniques.
Brush park	Submerged or partially submerged structures made of brush, branches, and aquatic vegetation secured in place with poles or fences. They are designed to provide fish rearing areas, fishing opportunity, or a refuge for fish. They vary in size from a few square meters to several hectares and are placed in lakes, streams, and brackish waters.

Catchment	See *watershed.*
Channelization	Straightening, narrowing, and deepening of a stream channel to improve navigation, move water faster, prevent flooding of human infrastructure, provide for construction of infrastructure, or other human uses. Often includes removal of debris and channel obstructions that may impede flow conveyance.
Channel unit	See *habitat unit.*
Coarse sediment	Generally, greater than 2-mm diameter, which is gravel, cobbles, or boulders. Compare *fine sediment.*
Control site	A study location nearly identical to the treated location, with the exception that no treatment occurs. See also *reference site.*
Coppicing	Forest regeneration from vegetative sprouts from stumps, branches, or roots.
Creation (habitat)	Construction of a new habitat or ecosystem where it did not previously exist. This is often part of mitigation activities.
Culvert	A transverse pipe or totally enclosed drain under a road or railway. Typically used to convey stream flow under a road or other manmade construction.
Deflector	One of many types of wood or stone structures placed perpendicular or at angle to a stream bank to deflect flow and create a pool and improve fish habitat or prevent a bank from eroding.
Easement	In restoration ecology, it refers to acquiring a portion of the rights to a land to allow for, or protect from, a specific use. Technically defined as the nonpossessory interest granted in the lands of another, established to obtain certain limited rights (e.g., development rights, but never the right to "quiet enjoyment") that are often in perpetuity but sometimes for only set periods of time. Compare *acquisition.*
Ecosystem	Dynamic and holistic system of all the living and dead organisms in an area and the physical and climatic features that are interrelated in the transfer of energy and material.
Effectiveness monitoring	Evaluating whether actions had the desired effects on physical processes, habitat, or biota. Compare *baseline, status, trend, implementation, and validation monitoring.*
Endangered species	Species in danger of extinction throughout all or a significant portion of its range. See also *threatened species.*
Enhancement	To improve the quality of a habitat through direct manipulation. It does not necessarily seek to restore processes or conditions to some predisturbed state. Some practitioners call this partial restoration. Compare *rehabilitation* and *restoration.*

Exclosure	Fencing an area to prevent (exclude) access of livestock or other ungulates.
Fine sediment	Generally, less than 2-mm diameter, typically composed of clay, silt, or sand. Compare *coarse sediment*.
Floodplain	A flat depositional feature of a river valley adjoining the channel, formed under present climate and hydrological conditions, and subject to periodic flooding.
Fry	Brief transitional stage of recently hatched fish that spans from absorption of the yolk sac through several weeks of independent feeding.
Gabion	Wire rectangular or round basket placed in a stream channel and filled with gravel, gobbles, boulders or other hard material to serve as bank protection or to create a weir to trap gravel, create a pool, and improve fish habitat.
Girdling	Removing the layer of bark and cambium around the circumference of a tree, usually performed in an attempt to kill the tree.
GIS (geographic information system)	A computer system for assembling, storing, manipulating, and displaying geographically referenced information.
Groin	A jetty extending from the bank into the channel designed to protect or stabilize a bank or trap gravel and sediment sands.
Gullying	Erosion of soil by formation or extension of channels (gullies) from surface runoff.
Habitat	In this report, the term refers to the aquatic environment that fish experience and not those landscape processes or attributes outside streams that alter habitat conditions.
Habitat unit	Distinct geomorphically defined area within a stream reach, such as a pool, a riffle, a glide, etc. Sometimes called a *mesohabitat*.
Hyporheic zone	The saturated interstitial areas below the streambed and into the streambanks (or floodplain), where stream water and deep groundwater intermix and where a number of important chemical, hydrological, and biological processes take place.
Implementation monitoring	Evaluating whether the restoration project was constructed (implemented) as planned. Compare *baseline, status, trend, effectiveness, and validation monitoring*.
Large woody debris (LWD)	Large piece of woody material such as a log or stump that intrudes into or lies entirely within a stream channel; LWD typically is defined as wood greater than 10 cm in diameter and 1 m in length, but other minimum size criteria also are used.

Levee	An embankment or dike constructed of earth, rock or other material to prevent a river from overflowing.
Limiting factor	Factor that confines (limits) the growth of an ecosystem element.
Long profile (longitudinal profile)	Longitudinal plot of elevation versus distance (gradient), typically along the *thalweg* of a stream.
Macroinvertebrate	Animal without a backbone, living one stage of its life cycle, usually the nymph or larval stage. Macroinvertebrates are visible without magnification, and many are benthic organisms (see *benthic*).
Main stem	Principle stream or channel of a stream network.
Mass wasting	Downslope movement of earth materials under gravity, including such processes as rockfalls, landslides, and debris flows.
Metapopulation	Network of semi-isolated populations, with some level of regular or intermittent migration and gene flow among them, in which individual populations may go extinct but then the habitat they occupied can become recolonized from remaining populations.
Metric	Standard of measurement or combination of parameters used as a standard of measurement. See also *parameter* and *variable*.
Mitigation	Action taken to alleviate or compensate for potentially adverse effects on an aquatic habitat that has been modified or lost by human activity.
Monitoring	Systematically checking or scrutinizing something for the purpose of collecting specific categories of data, especially on a recurring basis. In ecology, it generally refers to systematically sampling something in an effort to detect or evaluate a change or lack of change in a physical, a chemical, or a biological parameter. Compare *baseline, status, trend, implementation, effectiveness, and validation monitoring*.
N	Nitrogen
Nutrient enrichment	Addition of organic or inorganic compounds to a water body to increase background levels of nutrients (e.g., phosphorous, nitrogen).
Overstory	Uppermost layer of foliage that forms a forest canopy. Compare *understory*.
P	Phosphorus

Parameter	Quantitative physical, chemical, or biological property, such as water temperature or biota abundance, whose values describe the characteristics or behavior of an individual, a population, a community, or a ecosystem. See also *metric* and *variable*.
Parr	Young salmon during its freshwater residence between the fry and smolt stage and before the first seaward migration.
Periphyton	Sessile organisms, such as algae, that live attached to surfaces or rocks and other material projecting from the bottom of a freshwater aquatic environment.
Population	Group of individuals of a species living in a certain area that maintain some degree of reproductive isolation.
Power (statistical)	Probability of rejecting null hypothesis when it is, in fact, false and should be rejected. Typically denoted as alpha.
Primary production	Creation of organic matter (biomass) by photosynthesis or chemosynthesis.
Primary productivity	Rate at which organic matter (biomass) produced by photosynthesis or chemosynthesis is stored in an ecological community or group of communities. Compare *secondary productivity*.
Pseudoreplication	Use of inferential statistics to test for treatment effects with data from experiments, where either treatments are not replicated (though samples may be) or replicates are not statistically independent (S. H. Hurlbert, 1984, Pseudoreplication and the design of ecological field experiments, Ecological Monographs 54(2)187–211).
Reach	A geomorphically similar stream section or a section of stream as defined by two selected points.
Redd	Nest in gravel, dug by a fish for egg deposition, and associated gravel mounds.
Reference site	Site in a relatively natural state, representative of conditions before human disturbance. See also *control*.
Reclamation	Returning an area to its previous habitat type but not necessarily fully restore all functions.
Redd	A spawning nest constructed in the substrate of a lake or stream by a fish.
Rehabilitation	To restore or improve some aspects or an ecosystem but not fully restore all components. A general restoration term that can include habitat improvement, enhancement, or reclamation. Some

	practitioners call this partial restoration. Compare *enhancement* and *restoration.*
Remote sensing	Gathering data from a remote station or platform, as in satellite or aerial photography.
Restoration	1. Returning the ecosystem to some predisturbed condition. Some practitioners call this full restoration. 2. A general term for referring to various enhancement, improvement, and rehabilitation actions. Compare *enhancement* and *rehabilitation.*
Riffle	Shallow section of a river or stream, with moderate to rapid flow and with surface turbulence.
Riparian	The banks of a river or the terrestrial aquatic interface. That part of the terrestrial landscape that exerts a direct influence on stream channels or lake margins, and the water or aquatic ecosystems.
Riprap	Layer of large, durable materials such as rock used to protect a streambank from erosion; also may refer to the materials used, such as rocks or broken concrete.
Rubble matt	The placement of rocks, boulders, or concrete into the stream channel to create diverse flow and velocities and create riffles or fast water habitats for fishes and other aquatic organisms.
Salmonid	Fish of the family Salmonidae, including salmon, trout, and chars.
Samrah	See *brush park.*
Secondary productivity	Rate at which primary (plant and organism) material is synthesized into animal tissue per unit area in a given time period. Compare *primary productivity.*
Sediment supply	Supply of sediment to a river system, where it is carried in suspension (see *suspended load* and *wash load*) or on the bottom (see *bed load*).
Sheetwash	Surface erosion from water running off in sheets, distinct from channelized erosion in rills and gullies.
Side channel	A subsidiary or overflow channel branching from the primary stream channel, typically conveying a small fraction of the total stream flow.
Sill	A low weir partially buried in the stream bottom designed to aggrade or maintain the channel level and improve fish habitat.
Silviculture	In forestry or forest management, the care, cultivation, and harvest of trees. In restoration ecology, the term generally refers to planting, removing, or growing trees and other vegetation to restore certain forest characteristics.

Site	Place where a treatment or restoration activity may occur. Term can indicate a habitat unit or stream reach, depending on the coverage of the treatment or restoration activity.
Smolt	A young salmon just after the parr stage that has undergone the physiological changes to prepare it for its first migration from freshwater to the sea. A juvenile salmon in its seaward migration stage.
Snorkeling	A method for enumerating fish and aquatic organisms commonly used in clear lakes and streams that involved underwater visual observation using a mask and snorkel and other occasionally other diving equipment.
Soil creep	Gradual downslope movement of the soil mantle.
Staking and layering (riparian planting)	A form of riparian replanting that involves both the vertical placement of live shoots or cuttings and the horizontal placement of larger cuttings or branches. Commonly used for establishing willows to stabilizing banks and soils.
Status monitoring	Characterizing the condition (spatial variability) of physical or biological attributes across a given area. Compare *baseline, trend, implementation, effectiveness, and validation monitoring.*
Swale	A shallow troughlike depression that transports or retains water primarily during rainstorms or snow melts. They can either be natural topographic features or constructed depressions designed to retain water and prevent flooding.
Thinning	Removal of trees or other vegetation to allow for increased growth of other trees or vegetation.
Trend monitoring	Monitoring changes in biota or physical conditions over time. Compare *baseline, status, implementation, effectiveness, and validation monitoring.*
Understory	Shrubs and smaller trees between the ground cover and the forest canopy. Compare *overstory.*
Ungulate	Any of a number of hoofed, typically herbivorous, quadruped mammals, superficially similar but not necessarily closely related taxonomically.
Validation monitoring	Evaluating whether the hypothesized cause-and-effect relationship between restoration action or other treatment, and physical and biological response were correct. Sometimes considered a part of effectiveness monitoring. Compare *baseline, status, trend, implementation, and effectiveness monitoring.*

Water bar	Construction of small (generally less than 50cm) mound of soil and associated ditch across dirt or gravel roads to transport water across the road onto forest floor or adjacent hill slope. They are typically constructed on roads in mountainous areas to prevent water from running down the road surface and cause erosion.
Watershed	Entire land-drainage area of a river. Also called *basin, drainage basin,* or *catchment.*
Weir	1. A low dam constructed in a stream to divert or retain water for various human uses. 2. A low dam or obstruction constructed of logs or rocks placed across a stream to create a pool or trap gravel and improve fish habitat.
Wetted width	Width of the water surface within a channel. Contrast *bankfull width.*

This Fisheries Technical Paper is a synthesis of current information on inland fisheries habitat rehabilitation techniques. It summarizes the effects of various techniques for restoring natural processes, improving habitat, and increasing fish and other biota numbers and provides recommendations on effective techniques as well as information on planning and monitoring of rehabilitation projects. This document is intended to assist managers, field biologists and scientists involved in the planning, implementation and monitoring and evaluation of habitat rehabilitation and restoration of aquatic ecosystems.

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